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Environmental Assessment of Garden Waste Management

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DTU Environment
Department of Environmental Engineering

PhD Thesis
September 2009

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Preface

This thesis "Environmental Assessment of Garden Waste Management" is the result the PhD study conducted at the Department of Environmental Engineering of the Technical University of Denmark (DTU) from March 2006 to June 2009. The research project was supervised by Professor Thomas H. Christensen and involved practical work in the field, lab activities, modelling at the office and statistical evaluation of the results. Nine journal manuscripts relevant to this thesis were prepared during the course of the study and are enclosed as appendixes. They are referred to in the text by their roman numerals:

I. Boldrin, A. & Christensen, T.H. 2009. Seasonal generation and composition of Danish garden waste. Submitted to *Waste Management*.

II. Boldrin, A., Spliid, H. & Christensen, T.H. 2009. A novel approach for representative sampling of garden waste. Submitted to *Science of the Total Environment*.

III. Andersen, J.K, Boldrin, A., Christensen, T.H. & Scheutz, C. 2009. Mass balances and life cycle inventory for a garden waste windrow composting plant (Aarhus, Denmark). Submitted to *Waste Management & Research*.

IV. Boldrin, A., Andersen, J.K., Møller, J., Favoino, E. & Christensen, T.H. 2009. Composting and compost utilization: Accounting of greenhouse gases and global warming contributions. *Waste Management & Research*, 27, DOI: 0734242X09.

V. Andersen, J.K, Boldrin, A., Samuelsson, J., Christensen, T.H. & Scheutz, C. 2009. Quantification of GHG emissions from windrow composting of garden waste. Submitted to *Journal of Environmental Quality*.

VI. Boldrin, A., Hansen, T.L., Damgaard, A., Bhander, G.S. and Christensen, T.H. 2009. Modelling of environmental impacts from biological treatment of municipal organic waste (EASEWASTE). Draft manuscript for submission to *Waste Management & Research*.

VII. Boldrin, A., Hartling, K.R., Laugen, M.M. & Christensen, T.H. 2009. Use of compost and peat in growth media preparation: an environmental comparison using LCA-modelling (EASEWASTE). Submitted to *Resource, Conservation and Recycling*.

VIII. Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P. & Hauschild, M.Z. 2009. C balance, carbon dioxide emissions and global warming potentials in LCA-modeling of waste management systems. *Waste Management & Research*, 27, DOI: 10.1177/0734242X08096304.

IX. Boldrin, A., Andersen, J.K. & Christensen, T.H. 2009. Environmental assessment of garden waste management in the Municipality of Århus (EASEWASTE). Submitted to *Environmental Science & Technology*.

These papers are included in the printed version of the thesis but not in the www-version. Copies of the papers can be obtained from the Library at the Department of Environmental Engineering, DTU (library@env.dtu.dk).

In addition, the following publications have been produced during the Ph.D. study:

Christensen, T.H., Bhander, G.S., Lindvall, H.K., Larsen, A.W., Fruergaard, T., Damgaard, A., Manfredi, S., Boldrin, A., Riber, C. & Hauschild, M.Z. 2007. Experience with the use of LCA-modelling (EASEWASTE) in waste management. *Waste Management & Research*, 25, 257-262.

Boldrin, A., Andersen, J.K. & Christensen, T.H. 2009. LCA-report: Environmental assessment of garden waste management in Århus Kommune (Miljøvurdering af haveaffald i Århus Kommune). Department of Environmental Engineering, Technical University of Denmark.

Møller, J., Boldrin, A. & Christensen, T.H. 2009. Anaerobic digestion and digestate use: Accounting of greenhouse gases and global warming contribution. *Waste Management & Research*, 27, DOI: 0734242X09.

Boldrin, A. & Christensen, T.H. 2008. Life Cycle Inventory of peat in Denmark. Manuscript. Not published.

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Alessio

Summary

The amount of garden waste collected has been steadily increasing in Denmark during the last decade, representing a growing challenge for local authorities and waste companies. At present, most of the garden waste is treated in centralized composting facilities, but incineration and home composting have been recently proposed as alternative options.

The present thesis provides an environmental assessment of a range of treatment schemes for the management of garden waste, including windrow composting, thermal treatment and home composting. The results presented in this thesis are based on a study performed in Århus Kommune, which has financed most of the field activities performed. The activities included comprehensive field-sampling campaigns for characterization of garden waste and quantification of gaseous emissions during windrow and home composting.

During the waste characterization campaign, different properties of garden waste were defined, including unit generation rates, material fraction distribution and chemical compositions. Samples of waste were collected several times during the year, according to a low-cost sampling method developed and validated specifically for garden waste. The results confirmed that both material fraction and chemical compositions of the waste have a clear seasonal variability and suggest that diversion of garden waste to alternative treatments could be done both on single material fractions and on a seasonal basis. Furthermore, the chemical analyses showed that garden waste contains low level of pollutants.

Emissions of gases from outdoor windrow composting were quantified during several measurement campaigns and using different methods. The results showed that emissions of CH_4, N_2O, NH_3 and CO occur during composting, depending on the waste composition, the stage of the process, and the way the process is operated. Furthermore, it was concluded that small-scale measurement methods are not suitable for quantifying emissions from composting windrows because of the spatial and temporal variability of the emission pattern. A total-measurement approach was found more preferable for such purpose.

Both the results regarding the waste characterization and gas quantification campaigns where compared and validated with mass flow analysis (MFA) of the composting facility. MFA showed to be an essential tool both when calculating

waste specific emissions and when estimating transfer and degradation coefficients for the composting process.

Specific inventories for the technologies under study were established. The inventory for outdoor windrow composting of garden waste includes energy and material consumptions as well as environmental emissions defined for Århus composting plant. The inventory regarding home composting was established by using literature data and results from an ongoing experiment. The inventory of the waste-to-energy plant was obtained from a previous study, using data from Århus incinerator.

The Life Cycle Assessment (LCA) calculations were performed by means of the EASEWASTE model, accounting for all the relevant environmental aspects of garden waste management within a time horizon of 100 years. To fulfil the requirements of the assessment, the EASEWASTE model was further developed with two modules. The first module enables the user to estimate waste flows and environmental emissions – both process- and waste- specific – from biological treatment of organic waste systems. The second can be used for quantifying the environmental aspect of substituting peat with compost in commercial growth media. In addition, different accounting criteria for biogenic carbon contained in organic waste were analysed, concluding that different criteria are equivalent, but that correct and clear system boundaries definition is a prerequisite for obtaining consistent results.

In the Århus case study, six scenarios have been analysed, investigating waste diversion both on a material fraction and seasonal bases. The results are presented as a life cycle impact assessment (LCIA), including non-toxic (global warming, photochemical ozone formation, acidification and nutrient enrichment) and toxic-related impact categories. They show that a garden waste management system based on windrow composting generates smaller potential impacts than other types of waste and that utilization of compost in substitution of fossil growth media has important benefits for the environment. Some of the gaseous emissions occurring during windrow composting are among the major contributors to some of the non-toxic impact categories: CH_4 and N_2O to global warming, NH_3 to nutrient enrichment and acidification. Furthermore, the results reveal that the LCA methodology used for accounting of toxicity aspects related to heavy metals contained in soil might overestimate some of the calculated impacts, suggesting that proper adjustments of the methodology are needed.

The diversion of part of the waste to alternative options (incineration and home composting) could potentially lead to benefits for the system. In particular, introducing incineration could sensibly improve the GHG footprint of the garden waste management. However, the marginal technology for energy production seems to play a crucial role in such conclusion.

Dansk resumé

Mængden af indsamlet haveaffald har været jævnt voksende i Danmark i løbet af det sidste årti, og udgør hermed en stigende udfordring for de lokale myndigheder og affaldsfirmaer. På nuværende tidspunkt bliver størstedelen af haveaffaldet behandlet på centraliserede kompost faciliteter, men både forbrænding og hjemmekompostering er for nylig blevet foreslået som alternative muligheder.

Denne afhandling giver en miljøvurdering af en række forskellige behandlingssystemer for håndtering af haveaffald inklusiv milekompostering, forbrænding og hjemmekompostering. Resultaterne, som præsenteres i denne afhandling, er baseret på et studie foretaget i Århus Kommune, som har finansieret størstedelen af det udførte feltarbejde. Disse aktiviteter indbefattede omfattende prøvetagninger med det formål at karakterisere haveaffald og at kvantificere gasemissioner fra milekompostering og hjemmekompostering.

Under disse prøvetagninger blev forskellige egenskaber ved haveaffald bestemt, inklusiv den månedlige rate, fordelingen af materialefraktioner og den kemiske sammensætning. Affaldsprøver blev indsamlet adskillige gange i løbet af året ved brug af en prøvetagningsmetode med lave omkostninger, udviklet og valideret specifikt for haveaffald. Resultaterne bekræftede, at både materialefraktioner og den kemiske sammensætning af affaldet har en klar årstidsvariation, og indikerer at omdirigering af haveaffald til alternative behandlingsformer kan foretages - både på basis af de enkelte materialefraktioner så vel som årstidsbaseret. Endvidere viser de kemiske analyser, at haveaffald har et lavt indhold af forureningsstoffer.

Gasemissioner fra udendørs milekompostering blev kvantificeret under flere prøvetagningsrunder ved brug af flere forskellige metoder. Resultaterne viste, at emissioner af CH_4, N_2O, NH_3 og CO forekommer under komposteringen, og afhænger af affaldssammensætningen, de forskellige trin i processen og den måde, hvorpå processen styres. Endvidere blev det konkluderet, at punktmålinger ikke er egnede til at kvantificere emissioner fra komposteringsmiler grundet den rumlige og tidsmæssige variabilitet i emissionsmønstret. Det blev fundet, at et en tilgang baseret på total-måling er at foretrække til et sådant formål.

Resultater fra prøvetagningskampagner, både for affaldskarakterisering og kvantificering af gas emissioner, blev sammenlignet og valideret ved brug af materialestrømsanalyse (MFA) for komposteringsanlægget. MFA viste sig at være et essentielt værktøj – både til beregning af affaldsspecifikke emissioner og til estimering af transfer- og nedbrydningskoefficienter for komposteringsprocessen.

Specifikke opgørelser for de undersøgte teknologier blev fastlagt. Opgørelsen for udendørs milekompostering af haveaffald inkluderer energi- og materialeforbrug så vel som miljømæssige udledninger, defineret for Århus komposteringsanlæg. Opgørelsen for hjemmekompostering blev fastlagt ved brug af litteraturdata og resultater fra et igangværende laboratorieforsøg. Opgørelsen for forbrændingsanlægget blev fremskaffet fra et tidligere studie, som anvendte data fra Århus forbrændingsanlæg.

Livscyklusvurderingen (LCA) blev foretaget ved brug af EASEWASTE modellen, som redegør for alle relevante miljømæssige aspekter af haveaffaldshåndtering indenfor en tidshorisont på 100 år. For at opfylde alle betingelser for vurderingen blev yderligere 2 moduler udviklet i EASTEWASTE modellen. Det første modul muliggør at brugeren kan estimere affaldsstrømme og miljømæssige emissioner (såvel proces- som affaldsspecifikke) fra biologisk behandling af organiske affaldssystemer. Det andet modul kan bruges til at kvantificere det miljømæssige aspekt ved at substituere tørv med kompost i kommercielle vækstmedier. Derudover blev forskellige beregningskriterier for biogent kulstof i organisk affald analyseret, og det blev konkluderet, at nogle kriterier er ækvivalente, men at korrekte og klare definitioner for afgrænsningen af systemet er en forudsætning for at opnå konsistente resultater.

I det pågældende case study fra Århus blev seks scenarier analyseret, og det blev undersøgt hvad en omlægning af affaldshåndteringen, baseret på viden om materialefraktion og årstid, ville betyde. Resultaterne er præsenteret som en vurdering af potentielle livscyklus miljøpåvirkninger (LCIA), som inkluderer både ikke-toksiske påvirkninger (drivhuseffekt, fotokemisk ozondannelse, forsuring og næringssaltbelastning) og toksicitetsrelaterede påvirkninger. De viser, at systemer til forvaltning af haveaffald baseret på milekompostering genererer mindre potentielle påvirkninger end andre typer af affald, og at anvendelsen af kompost til substitution af fossilt vækstmedie har vigtige fordele for miljøet. Nogle af de gasemissioner, som forekommer under milekompostering, er blandt de største bidragydere til ikke-toksiske påvirkningskategorier: CH_4 og N_2O til global opvarmning, NH_3 til næringssaltbelastning og forsuring. Desuden afslører resultaterne, at LCA metodikken, som bruges til at redegøre for de toksiske aspekter i relation til tungmetaller i jord, muligvis overestimerer nogle af de beregnede påvirkninger, hvilket antyder at passende justeringer i metodikken er påkrævet.

Omdirigeringen af en del af affaldet til alternative behandlingsmetoder (forbrænding og hjemmekompostering) kan potentielt medføre fordele for systemet. Især kan indførelse af forbrænding medføre en væsentlig forbedring af drivshusgasudledningen for haveaffaldshåndtering. Dog lader det til, at den marginale teknologi til energiproduktion spiller en væsentlig rolle for en sådan konklusion.

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1. Background

1.1. Garden waste in Denmark

Garden waste is biodegradable waste consisting of different organic (e.g. grass clippings, hedge cuttings, tree prunings, small branches, leaves, and wood debris) and inorganic (e.g. soil, stones, and plastic bags) materials. Garden waste[1] is a low density and heterogeneous waste fraction generated during maintenance of public areas and private gardens. Factors such as season and geographical location (e.g. climate, urbanization, housing type, and waste management strategies) largely influence garden waste properties, determining its variable generation rates and composition.

Despite the fact that garden waste is becoming a significant waste stream in many countries, limited statistics are available regarding its generation at a European level. Garden waste is, in fact, in most cases collected mixed with food waste and data on generation rates for these two waste fractions are confounded (Eurostat, 2005). Conversely, garden waste has been separately collected in Denmark for a long period and dedicated statistics are available since 1994.

In 2006, garden waste collected in Denmark amounted to 598,000 tonnes, representing more than 18 % of municipal waste generation (Miljøstyrelsen, 2008). As shown in Figure 1, the amount of garden waste collected in Danish waste streams has been steadily increasing for more than a decade. This increase, however, does not represent an overall increase in waste generation, but it is rather a result of both new and more accessible waste collection schemes and improved collection of data regarding garden waste (Miljø- og Energiministeriet, 1999; The Danish Government, 2004).

The generation rate of garden waste has more than doubled in only 13 years, increasing from 67 kg person^{-1} year^{-1} in 1994 to 143 kg person^{-1} year^{-1} in 2006 (Boldrin & Christensen, I). Increasing garden waste generation is one of the major causes of increased generation of municipal waste generation in Denmark in the period 1994-2002 (Skovgaard *et al.*, 2005).

[1] also called yard waste or yard trimmings in American English.

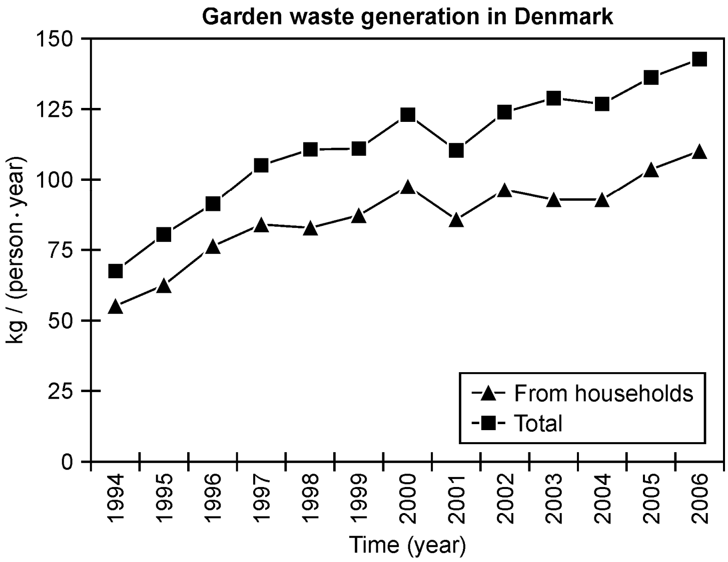

Figure 1 - Generation pro-capita of garden waste in Denmark in 1994-2006 (Boldrin & Christensen, I).

1.2. Treatment of garden waste in Denmark

Garden waste generated in Denmark is almost completely handled by means of one of two schemes: central composting and home composting[2]. Central composting treated 99% of the garden waste collected in 2006 (Miljøstyrelsen, 2008). To fulfil the increasing demand for treatment, the number of composting plants treating or co-treating garden waste in Denmark has been increased markedly in recent, from 110 in 1997 to 133 in 2001. More than 50 % of the capacity available in Danish composting plants is used for treating garden waste (Petersen & Hansen, 2003). Although the compost produced is increasingly being utilized by gardeners and professional landscapers, the most common use is still in private gardens (Petersen, 2001). Home composting of garden waste together with kitchen waste is also believed to be a commonly used option, but no precise statistics are available regarding the amount treated or the handling techniques (Petersen & Domela, 2003).

Central composting as a treatment option for garden waste builds on a long and well established practice rather than on a systematic evaluation of relevant alternatives, such as incineration and home composting. Incineration of garden waste in waste-to-energy (WTE) plants has been recently suggested as being

[2] also called backyard composting.

more environmentally friendly than central composting from a global warming perspective (USEPA, 2006). This conclusion is based on the consideration that utilization of increasing amounts of biomass can reduce the use of fossil fuels for energy production in the future, thus improving the carbon footprint of waste management systems. Incineration of garden waste in Denmark is only permitted in approved plants (The Danish Government, 2004). On the other hand, home composting of garden waste – possibly together with food waste - has been promoted in Denmark for a number of years through information campaigns and the results are considered very successful (The Danish Government, 2004). Further increase of home composting as an alternative to central composting has also been suggested, but such ideas should be carefully evaluated with respect to their potential environmental impacts, e.g. by means of LCA-modelling.

1.3. LCA and garden waste management

Life Cycle Assessment (LCA) is a tool developed for evaluating environmental aspects and potential environmental impacts occurring throughout the life cycle of a product. If properly adapted, the LCA principles can be applied to collecting and assessing data about the generation, collection and treatment of waste from an environmental perspective. The general principles and framework for LCA are described in the ISO-standard 14040 (2006).

The waste-LCA methodology is used for assessing different scopes and supporting different decision making levels. LCA can be used for environmental assessments regarding whole waste systems, the management of single waste material fractions, or the performance of specific treatment technologies (Christensen *et al*., 2007). Furthermore, depending on the aim of the study, LCA can be performed at different levels of detail: from simple screening for identifying general rules for waste management to a detailed and case-specific assessment in support of decision making at a local level.

Among others, the aims of an LCA study are to produce robust findings and to clearly convey recommendations practicable in “real life”. A waste-LCA study actually facilitates a decision making process only if the model, in itself, closely reflects the real system (in terms of flows and compositions), so that the results can be put into practice with a certain level of confidence. This can be achieved by properly scoping and interpreting the study, by using appropriate inventory data, and by implementing a modelling approach which adopts the necessary complexity and transparency. In addition, the discrepancy between the reality and the model can be reduced by accurately mapping all the flows of materials and substances occurring in the system (Christensen *et al*., 2007; Brunner & Ma, 2009).

The various (research) different activities detailed in this thesis covered all the different phases of an assessment identified in Figure 2, so that the LCA study could be uniformly improved in all its parts. Moreover, relevant information produced during each phase was used as feedback in the other phases. An overview of the assessed system was constantly provided, using Material Flow Analysis (MFA) and Substance Flow Analysis (SFA) to link rigorously and transparently the inputs and the outputs of treatment processes and management schemes.

This approach resulted in achievement of additional two goals. Firstly, the uncertainty inevitably affecting the results of LCA study was addressed methodically and reduced. In general terms, uncertainty affects a waste-LCA study with respect to the data used, the system definition, the modelling, the assumptions made, and the way results are interpreted. Secondly, the approach laid the basis for a systematic uncertainty analysis of the relevant aspects of the analysed system.

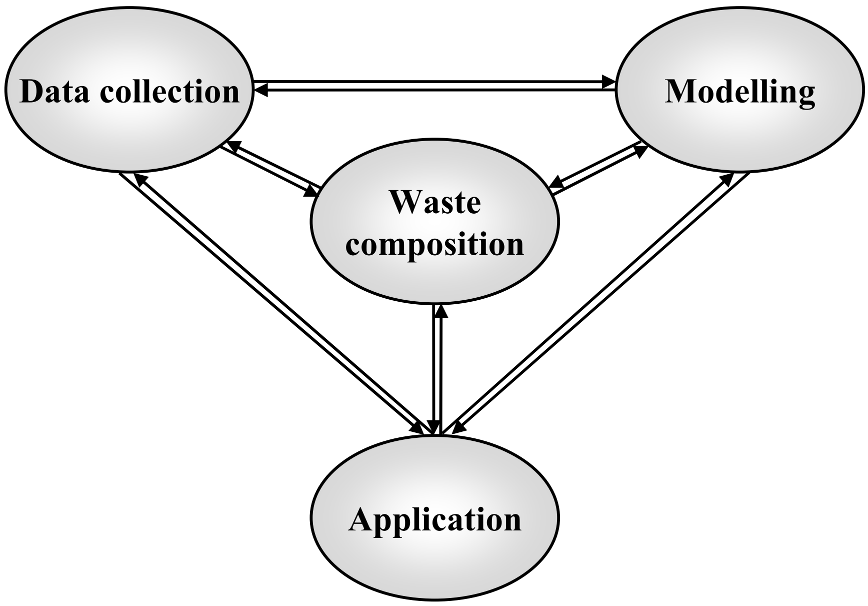

Figure 2 – Operative phases of an environmental assessment of organic waste management.

1.4. Aim of the thesis

The main aim of this Ph.D. thesis is to develop a basis for environmental assessment of garden waste management and provide data for this assessment in a Danish context. The presented approach and the LCA model are suitable for assessing other types of organic waste other than garden waste.

Based on this approach, a range of methodological and practical issues will be addressed and a description of relevant parameters and uncertain aspects in LCA of garden waste will be provided, in order to ensure complexity, specificity, flexibility, and consistency of the assessment. The thesis proves the importance of basing a LCA study on specific and accurate waste composition data and shows the potential for complementing LCA with Material Flow Analysis (MFA) modelling. Furthermore, it includes a first attempt to model in a LCA context the behaviour of private citizens when using compost in gardens.

The approach was adopted while carrying out an assessment for the Municipality of Århus. The study included an analysis and an environmental evaluation of the current garden waste management based on central composting and defined where improvements could be achieved in the system. Furthermore, alternative treatments, such as incineration and home composting, were assessed and the possibility of waste diversion on a seasonal basis was explored.

2. Garden waste composition

2.1. Waste composition data

In waste management, data about waste composition are used for different purposes: information about waste is indispensable, for instance, when designing treatment facilities (Burnley, 2007). From a management perspective, data about chemical composition of waste are relevant for assessing recycling, sorting and diversion schemes, for identifying the origin and occurrence of substances of environmental concern, and for defining the contribution of waste fractions to the overall material and substance flows in society (Riber *et al.*, 2009). Accurate material fraction and chemical compositions allow a precise and case-related determination of input-specific emissions and materials-energy consumptions during the inventory phase of an environmental assessment. Furthermore, an exact composition dataset is a prerequisite for MFA and SFA complementing the LCA study.

However, datasets regarding waste composition are often uncertain and seldom mutually comparable. Uncertainty is generated because the precision of a characterization campaign depends on the sampling method used (Gy, 1998), whereas most of the waste sampling methods have various inherent technical errors which reduce data quality (Dahlen *et al.*, 2008). The results from waste characterization campaigns are very difficult to compare because different studies present fundamental differences in sampling method (e.g. sorting prior or after mass reduction), classification and analytical procedures (Burnley, 2007; Riber *et al.*, 2007; Dahlen *et al.*, 2008). Uncertainty in the results could be reduced by statistically validating the sampling methods, while comparability could be improved by adopting standard sampling procedures. Technical, economical, and environmental constraints are obviously the limiting factors.

Sampling and characterization of garden waste can be difficult and costly because of its strong seasonality and heterogeneity. Garden waste composition has scarcely been covered in previous publications. Literature data were unsuitable for the aims of this Ph.D. thesis, because none of the reviewed studies reported both material fraction and chemical compositions of garden waste on a seasonal basis (Boldrin & Christensen , I).

2.2. Method for garden waste characterization

Methods for waste characterization can be classified in four categories (Brunner & Ernst, 1986; Riber *et al.*, 2009):

- Direct waste analysis (direct method);
- Market product analysis (material flows approach);
- Proximity study.
- Waste product analysis;

The choice of the correct approach depends on the scope of the study and the data needed. The operational procedure depends on local factors and practical issues such as sampling equipment, space availability, amount of waste to be sampled and economical constraints.

Direct waste analysis is probably the only method suitable for a complete characterization of garden waste. In fact, with regards to the other approaches:

- the market method is not applicable because garden waste is not a product traded on the market;
- proximity studies cannot be performed on garden since waste compositional data are not available;
- waste product analysis can only provide partial compositional information.

The direct method involves physical sorting, sampling, weighing and analysis of the waste. It can be performed at the source point (e.g. households) or at the treatment site (e.g. waste delivered at a composting plant). Sampling operations can be performed manually or by means of automatic mechanical equipment, with systematic or random sampling frequencies.

A main feature of garden waste is a seasonal variation in both unit generation and material fraction composition. Therefore, a comprehensive garden waste characterization necessitates repeated sampling campaigns throughout the year and the method employed must be capable of determining both the material fraction composition and the chemical composition of each of the fractions. A direct waste analysis method was planned and developed during this Ph.D. project and employed for a characterization campaign of garden waste. Material fraction and chemical compositions of garden waste were determined by collecting waste samples eight times during the year, twice per season, using all the waste received during the day of sampling as a primary lot in order to reduce the sampling error. The sampling method consisted of an initial sorting of garden waste into five material fractions (i.e. small stuff, branches, wood, foreign items, hard materials) followed by a 4-steps mass reduction scheme – industrial shredder, trailer shredder, 1-D sampling, riffle splitter –, according to representative sampling techniques (Boldrin *et al*., II). At each step, each fraction was subject to particle size reduction, mixing and sample splitting leading from initial sample sizes of more than 20 tonnes to a laboratory sample of few grams. Chemical analysis of the laboratory samples was performed by an external certified laboratory (ALS Scandinavia AB, Luleå, Sweden).

The method was validated in one of the sampling events by analysing 13 replicate samples of one of the material fractions (small stuff) collected according to a staggered sampling scheme and by examining the analytical results statistically. Analysis of Variance (ANOVA) was performed for all analytes and the Kolgomorov-Smirnov test was carried out for each of the mass reduction steps, concluding that mild uncertainty is introduced for the analyte "ash" at the top two levels (e.g. industrial shredder and trailer shredder). The sampling method was concluded to be robust and adequate for the scope of the characterization campaign (Boldrin *et al*., II).

2.3. Garden waste composition

Seasonal variation in generation rates, material fraction and chemical compositions of Danish garden waste was studied at the Århus composting plant, Denmark. Garden waste generation rates for Århus are shown in Figure 3 on a monthly basis. The figure shows clear and substantial variations in waste generation thorughout the year, with maximum amounts received during summer (around 20 kg person^{-1} month^{-1}) and minimum amounts during winter (around 3 kg person^{-1} month^{-1}). A secondary peak was recorded in autumn (October), probably due to the collection of leaves.

A clear seasonal dependence was found for both materials fraction and chemical compositions. A yearly weighted average material fraction composition (see Table 1) was calculated using information on the monthly generation rates (Figure 3) and the results of the sorting operations described above.

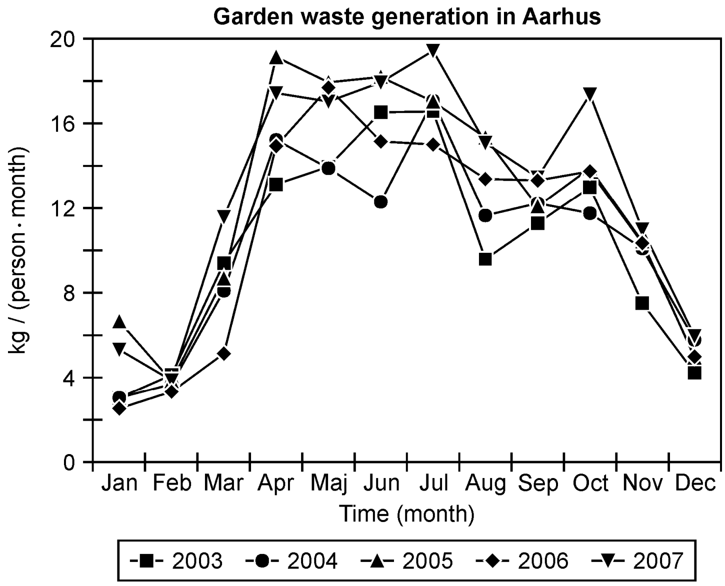

Figure 3 – Garden waste generation rates for Århus (Denmark) for 2003-2007 (Boldrin & Christensen, I).

Table 1 - Yearly weighted average garden waste material fraction composition (Boldrin & Christensen, I).

Waste fraction	Material fraction (%)
Small stuff	75.6
Branches	19.5
Wood	4.5
Stones	<0.2
Foreign items	<0.2
Total	100

Figure 4 shows the seasonal developments of Volatile Solids (VS), ash, and water contents, and Lower Heating Value (LHV). VS content and LHV are higher in winter months, as a result of the greater presence of woody material in the waste. The high ash content in summer is due to the significant amount of soil contained in "small stuff", which is the dominant waste fraction in this period of the year (Boldrin & Christensen, I).

The content of nitrogen is highest during summer and autumn months (Figure 5), when either grass clippings or leaves are the major constituents of garden waste. The C/N ratio changes considerably during the year, from around 30 in June to

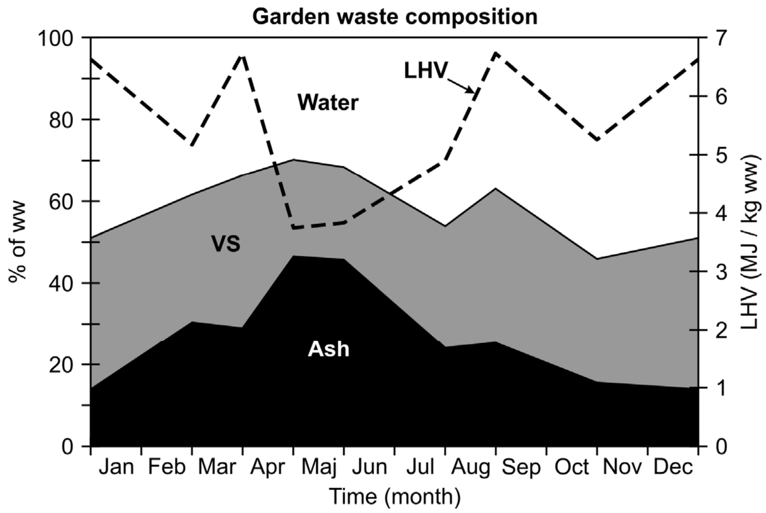

Figure 4 - Seasonal variation in water, VS and ash content, and LHV of garden waste (on ww basis) (Boldrin & Christensen, I). Please note different scales for y-axis. The dotted line represents LHV.

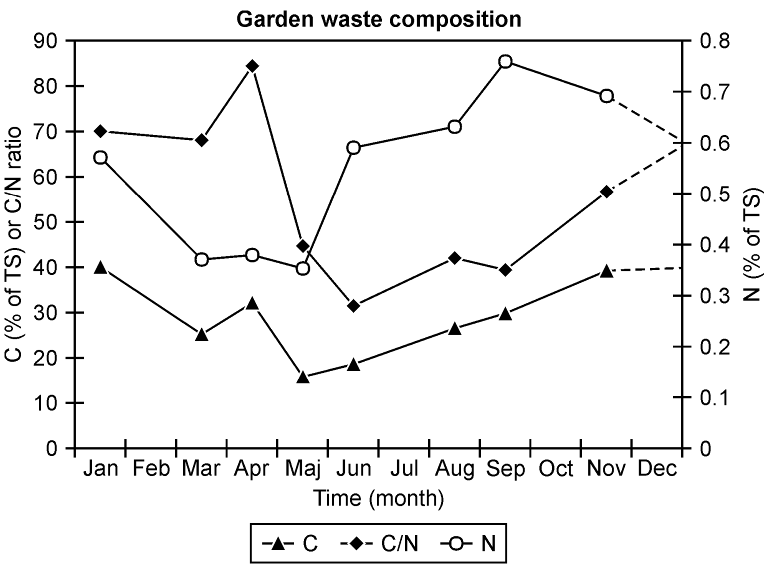

Figure 5 - Seasonal variation in carbon and nitrogen content, and C/N ratio of garden waste (Boldrin & Christensen, I). Please note different scales for y-axis.

more than 80 in late winter. Slightly different patterns were seen for P and K nutrients: P was found in highest concentrations in late summer (September), while K was prevalent in early summer (May and June). Correlation analysis was used to investigate the provenience of N, P, and K: a clear origin could not be determined for P, but both N and K were associated with the "small stuff" fraction. It was furthermore identified that N originates from both inorganic (i.e. soils) and organic materials, while P is mainly introduced by soil (Boldrin & Christensen, I).

Garden waste was also analysed for other trace elements of environmental concern (Cd, Cu, Hg, Pb, and Zn). Such analytes were all found in very low concentrations and spread across all material fraction, indicating that their presence is not regulated by specific mechanisms but it is rather accidental (Boldrin & Christensen, I).

2.4. Use of waste composition data

As presented later in chapter 3, the availability of data regarding generation rates, material fraction and chemical compositions of garden waste throughout the year made it possible to carry out a substance flow analysis of the system without it being biased by the variability of waste properties.

Moreover, the results of the characterization campaign suggested that the performance of alternative options for garden waste treatment could be investigated both from a material fraction standpoint and on a seasonal basis. For example, mixed garden waste collected in summer is not suitable for incineration, because of its high water and ash contents, and low LHV (Figure 4). In this case, incineration could be employed for the high calorific fractions (wood and branches), given that some sorting scheme is implemented.

On a seasonal basis, Figure 4 shows that garden waste collected in the winter months has suitable characteristics for incineration. In the same period of the year, the C/N ratio is not optimal for composting (Figure 5). These findings, combined with the fact that most of the recoverable N, P, K nutrients are contained in garden waste collected in summer months, suggests that composting and incineration could be to some extent alternated on a seasonal basis.

2.5. Uncertainty in waste composition data

No relevant sources of uncertainty in the presented garden waste composition can be indicated, both because the characterization method is considered robust and because no other complete studies are available in literature for comparison. However, it should be kept in mind that the data presented regards garden waste received at a specific treatment site, meaning that their suitability for more general studies should be considered carefully.

3. Collection and inventory of data regarding garden waste systems

3.1. Life Cycle Inventory (LCI) modelling

The Life Cycle Inventory (LCI) analysis of an LCA study involves modelling, compilation and quantification of inputs and outputs regarding a defined system. In some cases, the objectives of an LCA study can be achieved by compiling the LCI alone, without carrying out the impact assessment. A typical list of inputs normally includes the waste itself and the materials/energy necessary for its collection, handling, treatment and disposal. Outputs are defined as the products, materials, flows of energy and emissions leaving the system under consideration. If the system involves more than one waste flow, multiple products and different stages, allocation procedures might be needed.

Emissions occurring within a waste system usually fall into two categories: process-specific emissions and input-specific emissions (Bjarndottir *et al.*, 2002; Christensen *et al.*, 2007). Process-specific emissions depend on the amount of waste treated, on the treatment employed and on the way this process is operated. As a consequence, a specific technology will have the same emission regardless of the type of waste treated. Conversely, the magnitude and type of input-specific emissions are a function of the amount and composition of the waste flow in question; their accurate determination depends on the precision of the waste compositional data. The magnitude of input-specific emissions can give an indication of the relevant aspects to be determined regarding waste composition.

Data collection is carried out with respect to the goal and the scope of the study. Generic studies – used as basis for political decisions – or screening LCA normally make use of generic data reflecting, for instance, the typical technological level of a region or a country. For waste-LCA studies on particular waste systems, case-specific data fulfilling the technological scope are needed instead. The development of an inventory could, in this case, be an iterative process. An initial data collection gives an overview of the system so that new requirements or restrictions can then be identified. Based on this, new or extra data collection is performed. In some cases, a revision of the goal and the scope definition could be necessary.

Data included in the inventory are required to be appropriate for the scope of the assessment and representative for each impact category. The LCI must cover the temporal and spatial dimension of the scope (hereby avoiding assumptions), be functional to the level of complexity of the modelling and be consistent with previous findings. The use of specific and validated data definitely improves the credibility of the study by making it more realistic and reducing the gap between the modelled and the real system.

In some cases, parameter variability and uncertainty as well as inventory specificity represent the most important source of uncertainty in an LCA study (Huijbregts, 2002; Ross *et al.*, 2002). Data variability refers to spatial and temporal variations, and to variability related to the data source and object of the study (Björklund, 2002; Juijbregts, 2002; Heijungs & Huijbregts, 2004). Data uncertainty covers different aspects, such as inaccuracy, gaps, lack of representativeness, redundancy, and data modelling (Björklund, 2002: Heijungs & Huijbregts, 2004). The data accuracy and specificity could be improved with the adoption of appropriate and additional measurement methods, and by validating the results using mass balances and comparative analyses with similar studies (Björklund, 2002).

3.2. LCI for windrow composting of garden waste

Open-windrow composting is the most common composting technology in Denmark for handling garden waste: windrow composting was employed in 125 out of 133 plants treating garden waste in 2003 (Petersen & Hansen, 2003).

An inventory of data regarding windrow composting of garden waste has been established using Århus composting plant as a case study, supposedly representing the Danish situation. Good data quality and specificity of the assessment were insured by basing the inventory almost completely on measurements performed at the facility. Moreover, data were compared with literature findings to make sure they are consistent. A detailed description of the inventory analysis is reported in Andersen *et al.* (III).

3.2.1. Århus composting plant

In 2007, Århus composting plant treated 16,220 tonnes of garden waste generated in Århus Municipality, and delivered by trucks coming from recycling centres located in the municipality. Garden waste characteristics are those earlier described in paragraph 2.3. Garden waste undergoes sorting and shredding before being placed in windrows. The rows have a trapezoidal cross-section, they are 115 m long, 4.5 m high and 9 m wide, and they are turned every 6-8 weeks. Neither an aeration- nor an exhausts control system is installed at the facility. At the end of the composting period (55-60 weeks), the feedstock is double-screened: material of particle size <8mm is classified as compost and sold, woody materials in the range 8-25 mm are recirculated as structure material in the next batch of compost, while over-screen residues (>25 mm) are used as start-up material in the nearby WTE plant. A thorough description of the facility and the machinery used is reported in Boldrin *et al*. (2009).

3.2.2. Outputs

An average composition of composted material was determined by sampling the compost five times during one year. A large number of grab increments were taken from the pile of compost formed during the screening operations. The 70 kg sample was reduced to a 5 g laboratory sample by mean of a 2-steps mass reduction scheme. The first reduction was achieved by lying compost in elongated rows and taking out sub-samples using a perpendicular cross-cutting technique, similarly to that done for garden waste sampling (step 3). After drying of the compost material at 105°C, the final reduction was performed with a riffle splitter (Rationel Kornservice RK12, Esbjerg, Denmark). Physical-chemical analysis were carried out using the same procedure as described earlier for garden waste and reported in Boldrin *et al*. (II).

The results show that compost composition is constant throughout the year and it is therefore not influenced by the variable characteristics of the incoming waste. Furthermore, the compost produced at the Århus composting plant is a mature material and, due to its low content of pollutants, it is classified as suitable for organic farming (Andersen *et al*., III). On an annual basis, the amount of recoverable nutrients per tonne ww of compost is estimated to be: 5.14 kg of N, 1.25 kg of P, and 12.01 kg of K. Such values are in agreement with findings reported in other studies, as presented in Boldrin *et al*. (IV).

3.2.3. Material and substance flow analysis

Relevant flows of materials within the composting facility are shown in Figure 6. It can be seen that during the shredding operations, three fractions of material are sorted out and sent for treatment in different facilities: large items of wood (500 tonnes) to incineration, hard materials, e.g. stones (78 tonnes) to construction & demolition (C&D) waste recovery, and foreign items, e.g. plastic bags (106 tonnes) to incineration. No stocks of materials were recorded in the system.

Substance Flow Analysis (SFA) was carried out for several analytes for different purposes as reported in Andersen *et al*. (III). The ash balance was used for determining Total Solids (TS) transfer coefficients of the waste fractions into different outputs, while the C – and similarly VS – balance was used for determining the VS degradation ratios of the waste fractions during the composting process and for validating the measurements of C-containing gases.

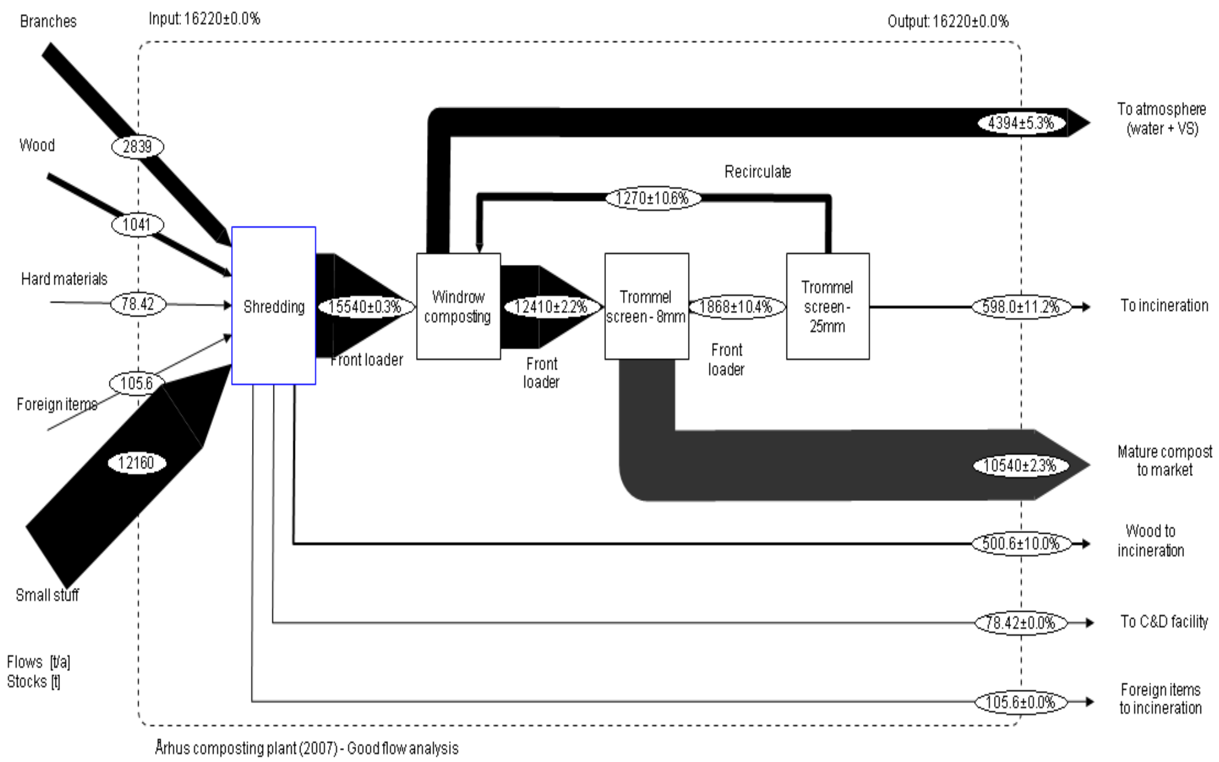

Figure 6 - Mass flow analysis (ww) of the Århus composting plant in 2007 (Andersen et al., III).

The aim of the mass balance for nitrogen (N) balance was to define the N degradation coefficients and to validate the measurements of N-containing gases, while the P flow analysis was intended to show that P is not degraded during composting. In both cases, the mass balances could not be resolved with great accuracy because of combined factors such as the low frequency of waste sampling and the variable contents in the waste input. However, the calculated uncertainties in N input and output indicate that N degradation (emitted to the air) does not exceed 10 % of the input. Within the reported uncertainty, the P flow analysis supports the hypothesis - commonly reported in literature (e.g. Eghball *et al.*, 1997; Kalamdhad & Kazmi, 2009) – that P is not degraded during composting.

The Cd and Cr flow analyses were chosen as representing the behaviour of most of the other trace elements, according to the correlation analysis presented in Boldrin & Christensen (I). Both balances could be resolved with an uncertainty of 10%, which is considered acceptable for MFA studies involving low concentrations and covering long time spans. In both cases it was concluded that leaching of heavy metals during composting is not occurring in relevant amounts.

3.2.4. Energy and materials requirements

Energy and materials are used in different activities performed during the composting operations. Energy requirements are in forms of fuel and electricity. According to Andersen *et al.* (III), in 2007, fuel (diesel) consumption in the Århus composting plant was 3.04 litres tonne^{-1} ww, with contributions from pre-treatments (i.e. shredding and piling of waste) and composting (i.e. turning with front loader and screening of mature compost). Electricity consumption was estimated to be 0.14 KWh tonne^{-1} ww - used in administration buildings, for the lighting of the composting area, and for heating the engines of the machinery in the morning.

Diesel and electricity consumption are in agreement with findings reported in other studies. As shown in Boldrin *et al.* (IV), typical fuel requirements for open technologies are in the range 0.4-6.0 litre tonne^{-1} ww, with a value of 3.0 litres being that most commonly reported. The Århus composting plant seems instead to be rather efficient with regards to electricity consumption. Values for electricity consumption in open technologies are, in fact, in the range of 0.023-19.7 kWh $tonne^{-1}$ of ww (Boldrin *et al.*, IV).

Other materials, required by the machinery during routine operations, were lubricating grease (0.013 l $tonne^{-1}$ ww), motor oil (0.005 l $tonne^{-1}$ ww), hydraulic oil (0.005 l $tonne^{-1}$ ww), and cleaning fluid (0.001 l $tonne^{-1}$ ww).

3.2.5. Emissions of CO_2, CH_4, CO, and N_2O

Gases emissions to the atmosphere and leachate release are potentially relevant burdens to the environment occurring during the composting process.

Gaseous emissions of CO_2, CH_4, CO, and N_2O are generated as result of microbial degradation of organic matter (Andersen *et al.*, III). Several field campaigns were conducted at the Århus composting plant in 2007-2008 in order to test different measurement methods, designed to investigate the formation of gases inside the windrows, and to quantify the magnitude of their emission in relation to the input material.

Gas formation was mapped by analysing compost air, sampled by means of gas probes driven into the windrows. Spatial variability was assessed using results from nine sampling points in the cross-section, while the temporal pattern in gas generation as a function of composting time was determined by sampling gas from windrows of different ages.

Spatial variability in gas formation is shown in Figure 7, where the average distribution of gas in compost air inside the windrows is reported. Each concentration is calculated as an average of 14 measurements taken from different windrows in two different events, as explained in Andersen *et al.* (V). Profiles for O_2 and CO_2 show that degradation of organic substrate is clearly occurring all over in the compost pile. However, high concentrations of CH_4 (and low O_2 level) indicate the occurrence of anaerobic conditions in the inner part of the windrow. This is very likely due to the large size of the compost rows and consequent high compaction of the feedstock, preventing sufficient diffusion of oxygen into the material. Formation of N_2O is, instead, more evenly distributed across the section, indicating that different processes concur in its production.

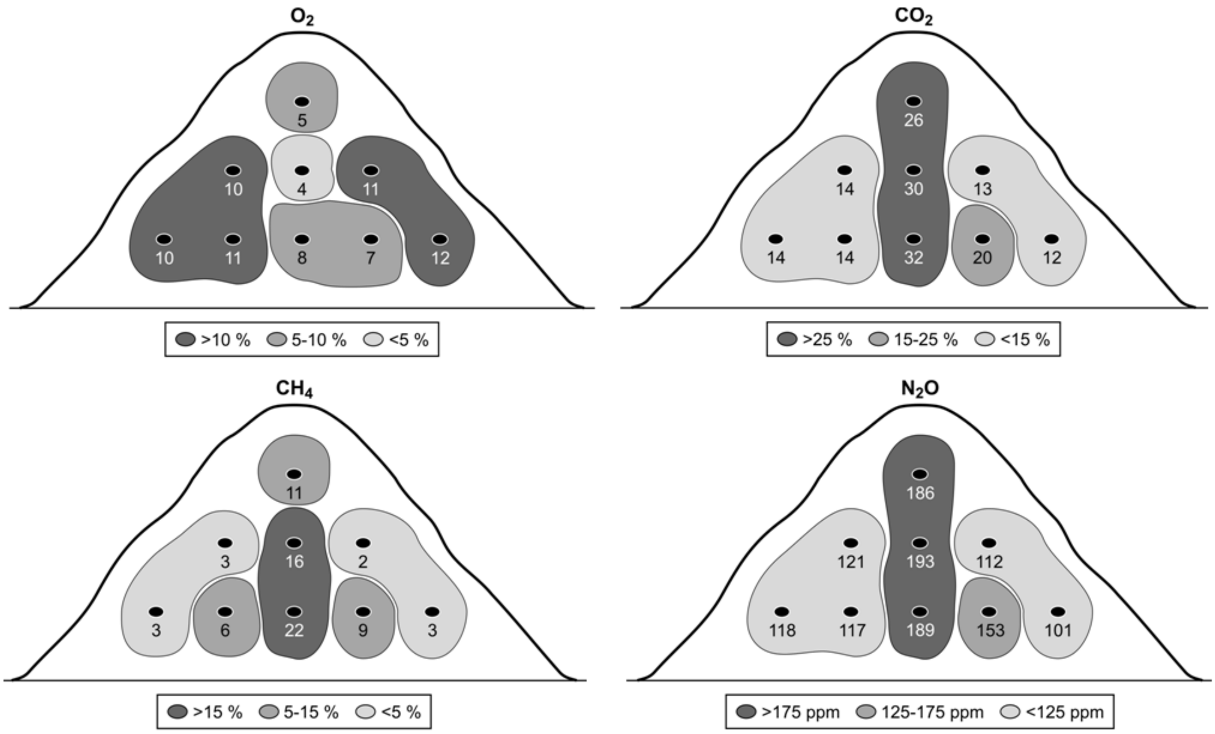

Figure 7 – Average cross-sectional distribution of O_2, CO_2, CH_4 and N_2O in compost air (Andersen et al., V). Please note different units.

Nitrous oxide (N_2O) is, in fact, a by-product of both nitrification and denitrification (Eggleston *et al*., 2006), primarily forming in anaerobic pockets where an oxygen gradient is present (Beck-Friis *et al*., 2000).

The degradation rate of organic matter varies during composting depending on different factors, including substrate and nutrients availability. Figure 8 and Figure 9 show typical developments in temperature and CO_2 production during composting of garden waste under laboratory conditions, where aeration was controlled to insure sufficient oxygen supply. The plots show an initial steep and swift increase of both parameters, indicating that the microbial activity is growing rapidly. A temporary steady state is seen at around 35°C, indicating a biological adaptation of the bacterial community to thermophilic conditions. A rapid increase follows and a peak is reached. Afterwards, because of less substrate availability, both parameters decrease and slowly approach background conditions (Boldrin, 2005).

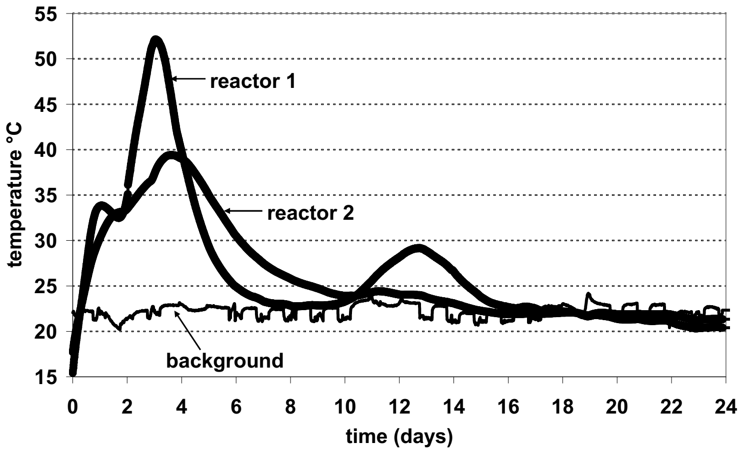

Figure 8 – Typical profiles for temperature during composting of garden waste in laboratory reactors (Boldrin, 2005).

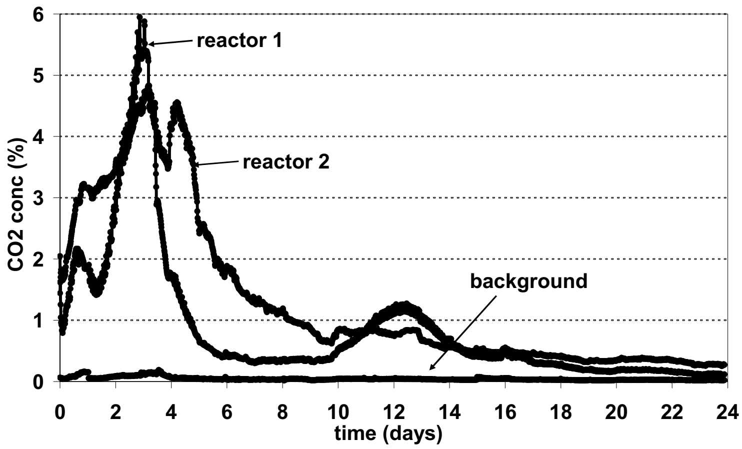

Figure 9 - Typical profile for CO_2 production during composting of garden waste in laboratory reactors (Boldrin, 2005).

Figure 10 shows the development of compost gas as a function of the material age in outdoor conditions, without forced aeration. Decreasing O_2 and increasing CO_2 and CH_4 concentrations suggest increasing biological activity and substrate degradation over time. However, concentrations varying by several orders of magnitude and the late peak of CO_2 reveal that other interfering factors (e.g. compaction) concur in determining such irregular patterns during the composting process compared to lab controlled conditions.

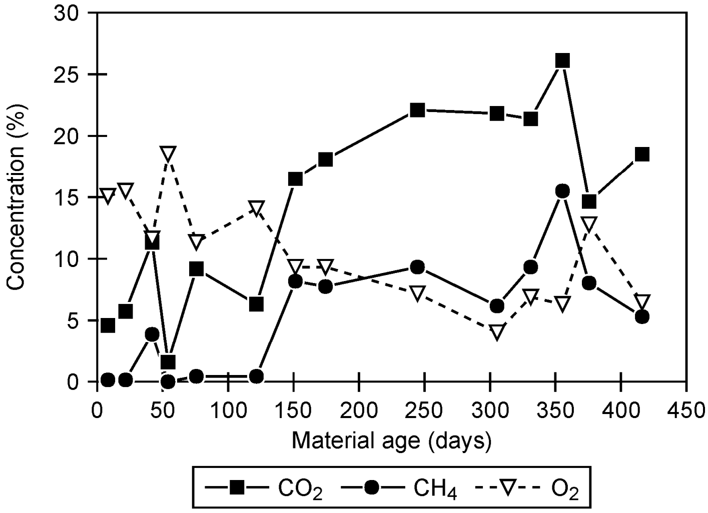

Figure 10 - Mean concentration of O_2, CO_2 and CH_4 in compost air during composting of garden waste as a function of composting age (Andersen et al., V).

The actual emission to the atmosphere of compost gases was investigated by means of a transport model and three measurement techniques estimating the flow (e.g. kg $hour^{-1}$) of gases leaving the composting rows. The transport model estimates gas emissions based on the compost air composition inside the windrows and using diffusion coefficients to model the transportation pattern. Two of the applied measurement techniques - flux chamber and dynamic funnel - are point-measurement methods, while the third method - dynamic plume - is a total-measurement method. Furthermore, the three measurement methods adopt different physical approaches. The flux chamber is a static method: the determination of the flux is based on the increase in gas concentrations in the chamber due to diffusion processes. The dynamic funnel and dynamic plume methods have instead a dynamic approach: the flux is determined by measuring the speed of the air flow and the (constant) gas concentration. The three systems and their functioning are thoroughly described in Andersen *et al*. (V).

Flow values obtained by way of the four methods (three measurement methods and the transport model) were “re-scaled” to estimate an annual release of gases from the composting windrows. As shown in Table 2, the methods generate rather different results.

Table 2 – Estimated annual release (tonne yr^{-1}) of gases from the Århus composting plant (Andersen et al., V).

Method	**Unit**	**CO_2**	**CH_4**	**N_2O**	**CO**
Flux chamber	tonne yr^{-1}	391 ± 109	21.4 ± 8.1	0.69 ± 0.11	0.18 ± 0.15
Dynamic funnel	tonne yr^{-1}	260 ± 86	10.1 ± 0.72	0.39 ± 0.14	0.17 ± 0.07
Dynamic plume	tonne yr^{-1}	6410 ± 769	50.2 ± 10.2	1.56 ± 0.72	5.53 ± 2.59
Transport model	tonne yr^{-1}	3663	508	2.86	0.67

The results in Table 2 were evaluated by comparing them with the carbon and nitrogen flow analysis results described in paragraph 3.2.3. The elemental mass of carbon and nitrogen contained in the gases under consideration was compared with the degradation estimates reported in the mass flow analysis. The results are reported in Table 3 and they show that the carbon balance could only be resolved with a reasonable accuracy by using the dynamic plume method and the transport model. However, the transport model estimated unrealistically high emissions of CH_4 compared to that measured in the compost air leaving the windrows. The reason for this is the fact that the transport model does not take into account oxidation of CH_4 to CO_2 in the aerobic layer situated just underneath the surface.

The point-measurement methods largely underestimate the emissions and are therefore not appropriate for the purpose of the study. Both the flux chamber and the dynamic funnel methods do not cover the observed high variability in emission pattern - several orders of magnitude over a very short time – due to the high dynamicity of the process (Andersen *et al.*, V).

The dynamic plume method was considered the most robust for the purpose of the assessment. Accordingly, values reported in Table 3 for the dynamic plume

Table 3 – Evaluation of the emission estimates for the methods under consideration (Andersen et al., V).

Method	**Emission (% of degraded element mass)**			
	CO_2-C	**CH_4-C**	**N_2O-N**	**CO-C**
Flux chamber	7.7 ± 2.2	1.2 ± 0.4	10.1 ± 1.6	0.006 ± 0.005
Dynamic funnel	5.1 ± 1.7	0.6 ± 0.04	5.7 ± 2.0	0.005 ± 0.004
Dynamic plume	127 ± 15	2.7 ± 0.6	23 ± 11	0.17 ± 0.08
Transport model	72	28	42	0.02

were adjusted to fit the mass balance and used as emission factors for the composting plant, according to Table 4. The estimated emission values are in line with the results of previous studies. As shown in Boldrin *et al*. (IV), for windrow composting of garden waste typical emissions of CH_4 are in the order of 2.1-2.7 % of degraded C, while N_2O emissions are in the range 0.5-1.8 % of input N.

Table 4 – Gas emission values for the Århus composting plant (Andersen et al., III).

Substance	**Unit**	**Århus**
Methane (CH_4)	% of degraded C	2.1
Nitrous oxide (N_2O)	% of total N	1.2
Carbon monoxide (CO)	% of degraded C	0.3

3.2.6. Emissions of NH_3

Quantification of NH_3 emissions from Århus composting plant could not be determined accurately. However, the various information available indicate that the emission is rather small.

First of all, the nitrogen balance (Figure 11) indicates that emissions of nitrogen containing gases are, in the worst case scenario (e.g. largest uncertainties included), around 10 % of the input N. Such low degradation of nitrogen during composting of garden waste with low frequency turnings was seen also in a laboratory experiment performed previously. In this experiment, batches of garden waste with different known C/N ratios (29.1, 42.5, and 52.25) were composted in reactors. Aeration was supplied for a short time and at different frequencies (9, 18, 27 days). After three aeration procedures, the reactors were emptied and the chemical composition of the output was analysed. As expected, the results (not published) show that with a C/N ratio over 50 the biological activity is low (Figure 12) and that relevant degradation of organic matter is only obtained with long a composting time. Furthermore, high C/N ratios (42.5 and 52.25) lead to low N loss (Figure 13). Under such conditions, the nitrogen necessary to the decomposing micro-organism for building cell structure is lacking and has to be recycled from the dying organisms, resulting in low ammonia emissions and slow decomposition of the organic matter.

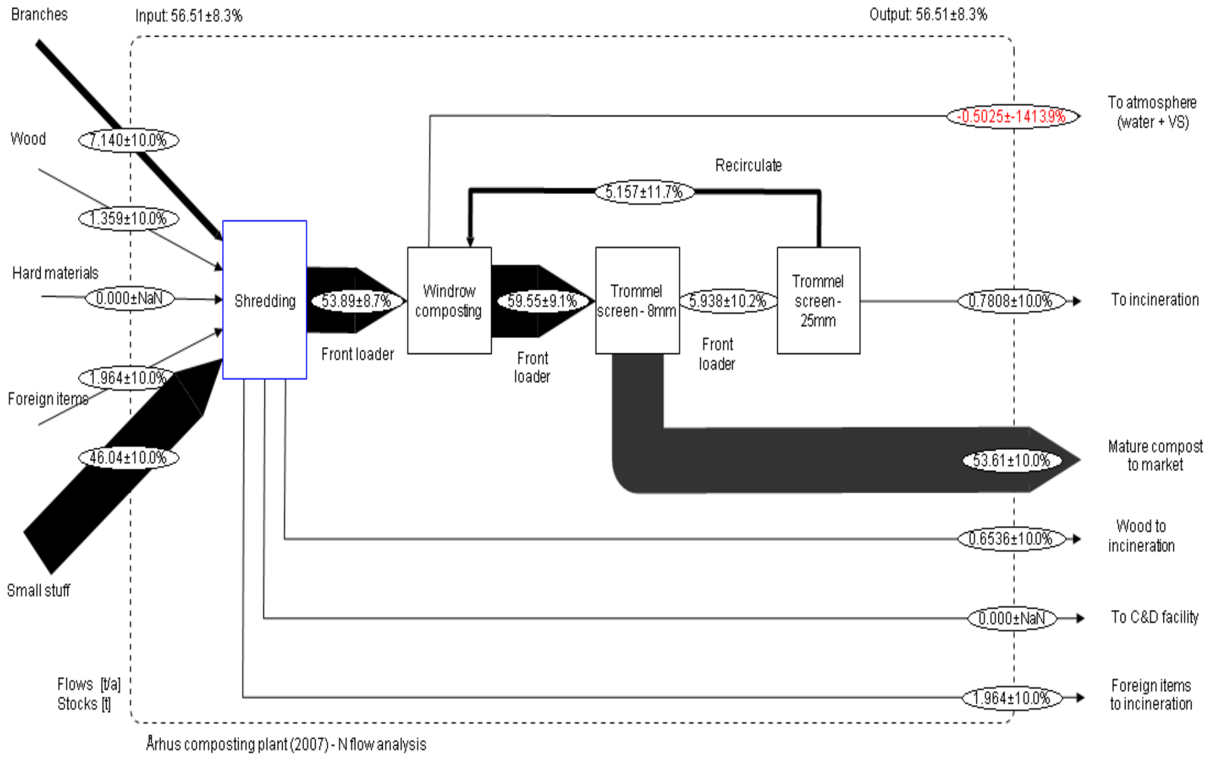

Figure 11 - Nitrogen flow analysis of the Århus composting plant in 2007 (Andersen et al., III).

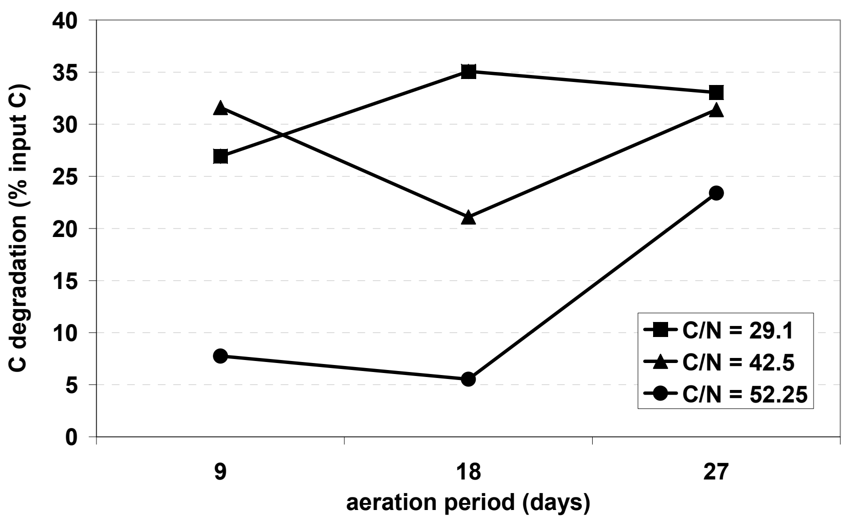

Figure 12 - Carbon degradation during composting of garden waste in laboratory reactors (data not published).

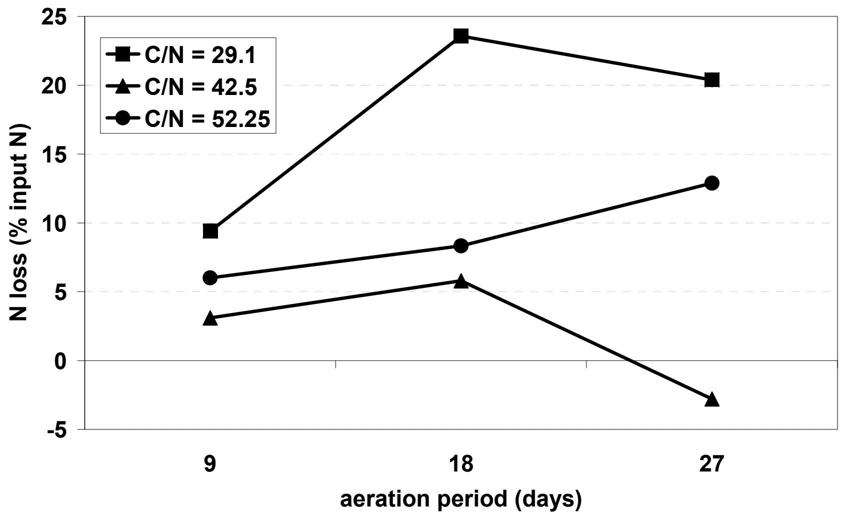

Figure 13 - Nitrogen loss during composting of garden waste in laboratory reactors (data not published).

Direct measurements at the composting plant have also been attempted, using both an active and a passive method. In the active method, gases were sampled from the dynamic funnel and forced into an absorbing solution by means of a pump. The sampling tube was placed in the central inner part of the funnel and the absorbing liquid solution comprised 10 ml of 0.02 N hydrochloric acid (HCl).

The procedure is described in detail in Liguori (2008). In the passive method, ammonia was absorbed on filters impregnated with 0.1 % citric acid and 1% glycerine solution, contained in diffusive plastic tube sampler with a length of 5.2 cm and a diameter of 1.2 cm. For a period of about nine days, samplers were placed in pairs in several locations on the top of the windrows - at about 20 cm from the surface – and at different distances from the composting pad, in order to assess both the emissions of NH_3 from the composting rows and the dispersion/deposition in the area surrounding the composting facility. For both methods, concentrations of NH_3 were determined by spectrophotometry (Liguori, 2008).

The results from both methods show that concentrations of NH_3 emitted from the windrows are very low. For the active method, all samples had concentrations below the detection limit of the spectrophotometer (10 μg/l or 10 ppbv). The passive method confirmed that most of the samples had a content of NH_3 below 10 ppbv. However, some of the samples placed over the youngest windrows showed higher concentration. The difference between the two methods is explained by the different approaches. The results of the active method are based on short term measurements and therefore their reliability is largely affected by the variability of the NH_3 fluxes from the windrows, which, in turn, depends on different factors such as weather condition, wind, and temperature. The results of the passive method represents, instead, an average concentration over a certain period (nine days in this case) and variable conditions are averaged out.

Furthermore, according to the results shown in Figure 14, the passive method gave indications of the emission pattern for NH_3. Firstly, the NH_3 emissions are higher in the young windrows and decrease with increasing compost age. Secondly, measured concentrations of NH_3 outside the composting area are within typical background levels for rural areas (e.g. 1-6 ppbv) indicating that the composting facility is not an important source of gaseous NH_3.

Combining the different information available, it was decided to assume an N degradation value corresponding to 8% of the input N in the inventory analysis. This is a cautious estimate, similar to what reported in Amlinger *et al*. (2003) for mixed organic waste - with a higher N content and a lower C/N ratio – and

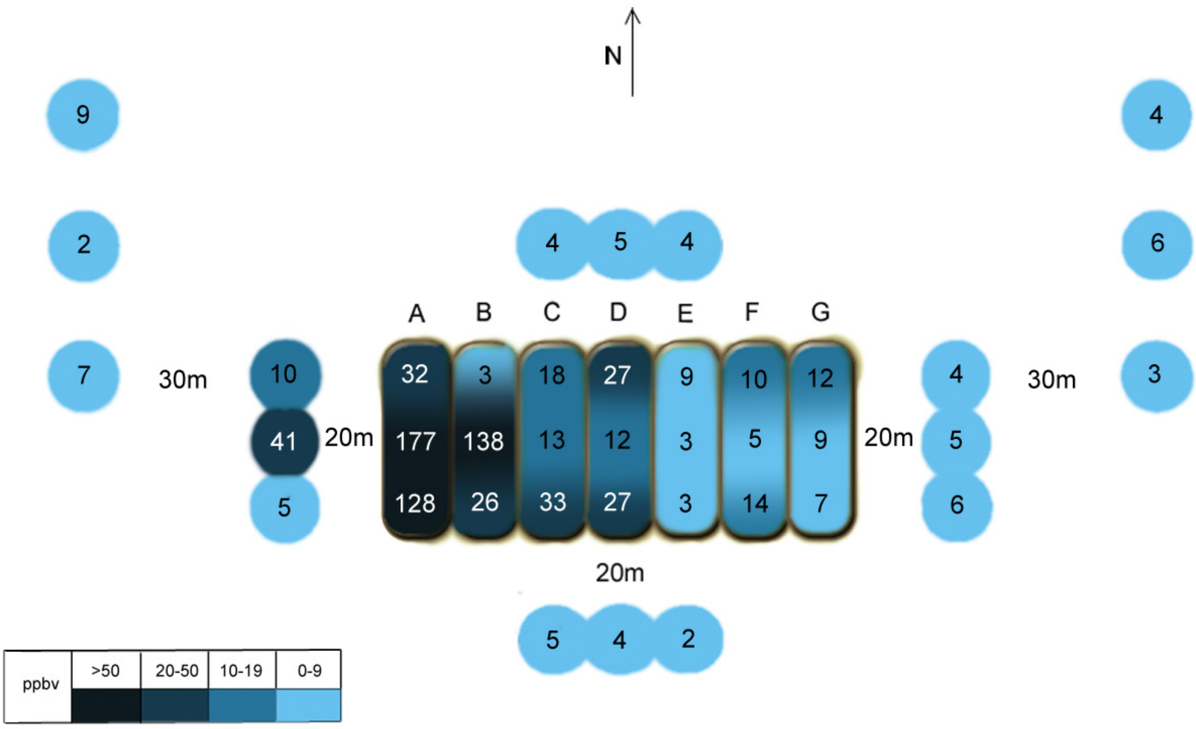

Figure 14 – Distribution of NH_3 concentration (in ppbv) above composting windrows (from A to G) and in the area surrounding the composting facility (Liguori, 2008).

higher than the 1.2 % reported in Hellebrand (1998) for windrow composting of garden waste. As reported in the literature, degraded nitrogen is almost entirely emitted as NH_3 (Dalemo *et al*., 1997; Beck-Friis *et al*., 2001; Ham & Komilis, 2003; Den Boer *et al*., 2005). Since N_2O represents 15% of the degraded N (1.2 % of input N), it is assumed that ammonia accounts for 83 % of the nitrogen losses and that the remaining 2 % is emitted as N_2.

3.2.7. Liquid emissions

Wastewater generated in the composting facility is collected in the sewage system. Considering that the facility comprises of 6000 m^2 of paved area and 7000 m^2 of storage buildings, and that the mean precipitation is 500 mm yr^{-1}, the amount of wastewater was estimated to be around 6500 m^3 $year^{-1}$. Whether leaching is adding an extra amount of wastewater could not be determined. However, leaching from windrows is believed to be limited, because the low moisture content and the high temperature in the piles will cause initial absorption and later evaporation of rain rather than leaching.

3.3. LCI for incineration of garden waste

A data inventory regarding thermal treatment of garden waste was compiled using data from the Århus WTE plant. As described in detail in Riber *et al.* (2008), the Århus WTE plant employs a furnace grate technology with Combined Heat and Power (CHP) energy recovery system. The annual capacity of the plant is 240,000 tonne and the energy production efficiency – calculated on the LHV - is 20.7 % for electricity and 74 % for heat. Cleaning of flue gas is performed with semidry (two lines) and wet (one line) systems. Dioxins and Hg are removed by means of activated carbon, while a Selective Non-Catalytic Reduction (SNCR) system is used for NO_x removal. The inventory includes different inputs of materials and energy into the process.

The Århus WTE plant can be considered to some extent as an example of a typical Danish WTE facility. The flue gas cleaning system is based on commonly employed technologies (Danish Energy Authority, 2005). The energy recovery efficiency is a slightly higher than the average efficiency of Danish WTE plants -

16% for electricity and 68% for heat (Energistyrelsen, 2007) – but lower than latest generation of facilities installed (Danish Energy Authority, 2005).

3.4. LCI for home composting of garden waste

Home composting is typically performed in small bins or containers placed in private backyards. Input material for home composting is normally food waste and small-size garden waste. Home composting can be considered as a non-controlled process, where monitoring of parameters such as moisture content, chemical composition, and temperature is not carried out. A "typical" home composting process is not definable, as each house owner manages the process in a different way.

Home composting is scarcely covered in the literature and very few studies including process data and emissions to the environment can be found. Data mining becomes even more problematic if the focus is solely on garden waste.

A long term experiment on home composting is ongoing at DTU (Danish Technological University). The aim is to define the environmental profile of home composting, including gaseous emissions, leaching from the composters, and degradation coefficients for organic matter for the different waste fractions. Moreover, the experiment intends to simulate different handling conditions, such as mixing frequencies and waste addition rates. The experiment is in its final phase, but only some preliminary results were available at the time of writing this thesis. These results were used to support and compare literature data.

For the present LCI, gas emissions were estimated as shown in Table 5 according to Amlinger *et al*. (2008), supported by the preliminary results of the experiment. Based on the carbon degradation derived from these emissions factors, it was assumed that the VS degradation corresponds to 40% of the input VS. The N

Table 5 – Estimated emission values for home composting (from Amlinger et al., 2008).

Substance	**Unit**	**Emission value**
Methane (CH_4)	% of degraded C	3.0
Nitrous oxide (N_2O)	% of total N	1.05
Ammonia (NH_3)	% of total N	6.3
Carbon monoxide (CO)	% of degraded C	0.04

degradation was assumed to be 7.5 % of the input N, 84 % of which emitted as NH_3 and 14 % as N_2O. No data regarding leaching from the composters were available and therefore leaching values are not included in the inventory.

3.5. Use of inventory data

From an LCA standpoint, LCI data fed to LCA models can be responsible for a large share of the uncertainty. The data collection presented was planned and carried out to obtain specific and robust datasets. The above described activities showed that a detailed inventory, including both process- and input-specific emissions and covering the temporal and spatial scope of the assessment, can be established with a reasonable effort.

The use of MFA for the modelling of the LCI was important for at least three reasons. Firstly, the presented mass balances showed that input-specific emissions could be determined with a certain degree of accuracy. Such accuracy depends, however, on access to a precise waste composition dataset. Secondly, MFA provided the basis for estimating transfer coefficients for the different waste fraction and degradation values for VS. Thirdly, the description of the system reported in the mass flows helps the LCA modelling (chapter 4) in reflecting the real studied system.

The LCI analysis of the Århus composting plant represents the first attempt to quantify gaseous emissions from an outdoor full-scale facility and indicates that reliable data, not biased by spatial and temporal variability, can be obtained employing a total-emission method. The results confirm previous findings reported in literature and definitely allow the conclusion that emission of greenhouse gases are significant during composting and should not be neglected in LCA studies. This means that the assumption that no CH_4 is emitted during composting (Smith *et al*., 2001; USEPA, 2006; Recycled Organics Unit, 2003; Cabaraban *et al*., 2008) and not including N_2O emissions in the accounting (Smith *et al*., 2001; Recycled Organics Unit, 2003) are both incorrect and could potentially bias the results of the assessment.

3.6. Uncertainty in data inventory

Different aspects contribute to the uncertainty in the inventory presented above. The inaccurate data on nitrogen content and missing data regarding NH_3 emissions lead to an uncertain nitrogen balance, which could bias the modelling of gas emissions and recovery of nutrients. The absence of other studies, quantifying NH_3 emissions from a full-scale facility, prevents any thorough comparison with existing data.

The LCI of home composting also suffers from gaps in both measured and literature data. Such gaps are bridged partly with estimated data or with preliminary results of ongoing experiments. An aspect which has instead been excluded from the inventory due to lack of information is the construction and decommissioning of treatment facilities. This issue could represent a model uncertainty, biasing to some extent the environmental assessment.

4. Modelling of data and impacts from garden waste systems

4.1. Waste-LCA modelling

Environmental waste-LCA is a system analysis tool for studying environmental aspects and potential impacts arising throughout the life cycle of waste. A waste-LCA model includes both a framework useful for defining and calculating material flows and energy turnovers in waste management systems and an evaluation toolkit, normally based on LCA-methodology (Dalemo *et al*., 1997; Kirkeby *et al*., 2006a).

Like all models, a waste-LCA model is a "simplified characterization of some aspects of the reality" (Norris, 2009). A model must therefore be complex – to include as many aspects as possible – and simple – so that transparent information is accessible and understandable, also to non-expert user (Ekvall *et al*., 2007). Furthermore, to assure a life-cycle perspective to the assessment, the model must include both direct activities regarding waste handling and those activities which are external to the waste system but are linked to it. Couched in other terms, the waste-LCA model must permit system boundary expansion, which is useful when assessing multi-functional systems and for avoiding intricate allocation procedures.

As introduced earlier (chapter 3.1), a waste-LCA model needs to accommodate both waste-specific[3] and process-specific emissions. In particular, waste-specific emissions necessitate that input and output compositions are linked and that different compounds contained in the waste are tracked by means of Transfer Coefficients (TC).

Several waste-LCA models for the assessment of waste systems are available, differing from each other by way of a number of technical assumptions and solutions. Major differences are also inherent in the level of flexibility and the quality of the database they provide.

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[3] also called waste-related or input-specific emissions

Different factors contribute to uncertainty in LCA-modelling. First of all, different factos such as data availability, allocations, geographical scale, temporal horizon and differentiation, suitable characterization factors, and knowledge of the analysed system normally affect the correctness of the system-boundary definition and the robustness of the modelling. Secondly, errors occur when modelling the process tree – also called the chain of processes - for including upstream and downstream processes (system expansion), because of assumptions made concerning the supplying chain and allocations (Heijungs & Huijbregts, 2004; Ekvall *et al*., 2007; Norris, 2009). Another aspect is represented by truncations of "negligible" flows and processes, performed in order to simplify the modelled system, but enlarging the gap between the modelled and the real systems. Finally, the impact assessment method could represent a source of error, because spatial, temporal, threshold and dose response information are typically excluded or aggregated over time (Ross *et al*., 2002).

4.2. EASEWASTE modelling

The EASEWASTE waste-LCA-model was developed for estimating waste flows, resource consumption and environmental emissions from waste management systems. A complete potential impact assessment within a 100-year time horizon can be carried out, embracing a range of potential impact categories, such as global warming, ozone depletion, photochemical ozone formation, acidification, nutrient enrichment, ecotoxicity and human toxicity. The general concept of EASEWASTE is described in Kirkeby *et al.* (2006a).

EASEWASTE operates with up to 48 waste material fractions, each one described by 40 components, and includes several modules for modelling specific technologies for handling, treating, or disposing of waste.

Assessment of alternative options for garden waste management involves the modelling of different treatment technologies and the evaluation of the utilization of the residues generated. As introduced earlier, the technologies taken into consideration are (central) windrow composting, home composting and incineration. The first two are modelled in the so-called "Biotechnology" module of the EASEWASTE model, while the latter is defined in the "Thermal Treatment" module.

4.2.1. Biological treatment modelling

The modelling of biological treatments in the EASEWASTE model is described in detail in Boldrin *et al*. (VI). The biological degradation of organic matter (VS) during composting is modelled for each material fraction defined in the waste composition table by means of VS degradation ratios. Assuming that carbon (C) degradation is proportional to VS degradation, the amount of carbon [kg] degraded during composting and emitted to air (C_{air}) is calculated according to Equation 1 (Boldrin *et al*., VI):

$$C_{air} = M_t \times \sum_f \left(m_f \times TS_f \times C_f \times VS_{\deg,f} \right) \qquad \text{Equation 1}$$

Where:

M_t	[kg]	is the input wet waste mass
m_f	[%]	is the mass fraction of the material fraction –f in the waste
TS_f	[% ww]	is the dry matter content of fraction –f
C_f	[% TS]	is the carbon content of fraction –f
$VS_{deg,f}$	[%]	is the VS degradation ratio for fraction –f (user defined)

For windrow composting, the VS degradation values for each of the garden waste fractions described in chapter 2.4 have been estimated by combining the mass balances for VS and ash. For home composting, no information was available, accordingly an assumption (40 % degradation) was made. The results are reported in Table 6.

Table 6 - VS degradation coefficients during composting of garden waste.

Fraction	VS degradation (% VS input)	
	Windrows composting	Home composting
Small stuff	77	40
Branches	28	-
Wood	12	-

4.2.1.1. CO_2 emissions from composting

The amount of CO_2 emitted is linked to the amount of degraded C (C_{air}) previously calculated minus the fraction of carbon emitted as CH_4. In addition, CO_2 is generated by methane oxidation in a biofilter, if present. The amount [kg] of CO_2 emitted to atmosphere ($CO_{2,air}$) is calculated according to Equation 2 (Boldrin *et al.*, VI):

$$CO_{2,air} = C_{air} \times \left(\left(1 - CH_{4,degr\_C}\right) + \left(CH_{4,degr\_C} \times CH_{4,clean}\right)\right) \times \frac{molar\_CO_2}{molar\_C} \quad \text{Equation 2}$$

Where:

C_{air}	[kg]	amount of C degraded and emitted to the air (Eq. 2)
$CH_{4,degr\_C}$	[%]	fraction of C_{air} emitted as CH_4
$CH_{4,clean}$	[%]	CH_4 removal efficiency in the biofilter
$molar\_CO_2$	[g mole^{-1}]	molar weight of CO_2
$molar\_C$	[g mole^{-1}]	molar weight of C

In windrow composting and home composting a biofilter is not present and thus $CH_{4,clean}$=0. The previous equation becomes therefore:

$$CO_{2,air} = C_{air} \times \left(1 - CH_{4,degr\_C}\right) \times \frac{molar\_CO_2}{molar\_C} \quad \text{Equation 3}$$

4.2.1.2. CH_4 emissions from composting

The amount of methane released to the atmosphere is defined as a percentage of the degraded C. If a biofilter is installed, CH_4 removal efficiency is also to be specified. The amount of CH_4 [kg] emitted to the atmosphere ($CH_{4,air}$) is calculated according to Equation 4 (Boldrin *et al.*, VI):

$$CH_{4,air} = C_{air} \times CH_{4,degr\_C} \times \left(1 - CH_{4,clean}\right) \times \frac{molar\_CH_4}{molar\_C} \quad \text{Equation 4}$$

Where:

C_{air}	[kg]	amount of carbon degraded and emitted to air (Eq. 2)
$CH_{4,degr\_C}$	[%]	fraction of C_{air} emitted as CH_4
$CH_{4,clean}$	[%]	CH_4 removal efficiency in the biofilter
$molar\_CH_4$	[g mole^{-1}]	molar weight of CH_4
$molar\_C$	[g mole^{-1}]	molar weight of C

Also in this case it is $CH_{4,clean}$=0 for both windrow and home composting, and therefore Equation 4 becomes:

$$CH_{4,air} = C_{air} \times CH_{4,degr\_C} \times \frac{molar\_CH_4}{molar\_C} \qquad \text{Equation 5}$$

As reported in chapter 3.3, the CH_4 emission coefficients have been estimated to be $CH_{4,degr\_C}$ = 2.2 % for windrow composting and $CH_{4,degr\_C}$ = 3.0 % for home composting.

4.2.1.3. N emissions from composting

The amount of nitrogen [kg] degraded during the composting process and emitted to the atmosphere (N_{air}) is calculated according to Equation 6 (Boldrin *et al.*, VI):

$$N_{air} = M_t \times \sum_f \left(m_f \times TS_f \times N_f \times N_{\deg,f} \right) \qquad \text{Equation 6}$$

Where:

M_t	[kg]	is the input wet waste mass
m_f	[%]	is the mass fraction of the material fraction –f in the waste
TS_f	[% ww]	is the dry matter content of fraction –f
N_f	[% TS]	is the nitrogen content of fraction –f
$N_{deg,f}$	[%]	is the N degradation ratio for fraction –f

As explained in chapter 3.3, $N_{deg,f}$ is estimated to be 8% and 7.5 % of the input N for windrow composting and home composting respectively. Emissions of nitrogen-containing gases are then divided into NH_3 ($NH_{3,air}$) and N_2O (N_2O_{air}), according to Equation 7 and Equation 8 (Boldrin *et al.*, VI):

$$NH_{3,air} = N_{air} \times NH_{3,\deg r\_N} \times \left(1 - NH_{3,clean}\right) \times \frac{molar\_NH_3}{molar\_N} \qquad \text{Equation 7}$$

$$N_2O_{air} = N_{air} \times N_2O_{\deg r\_N} \times \left(1 - N_2O_{clean}\right) \times \frac{molar\_N_2O}{molar\_N} \qquad \text{Equation 8}$$

Where:

$NH_{3,clean}$	[%]	removal efficiency in the biofilter
N_2O_{clean}	[%]	removal efficiency in the biofilter
$NH_{3,degr\_N}$	[%]	fraction of N_{air} emitted as NH_3
$N_2O_{degr\_N}$	[%]	fraction of N_{air} emitted as N_2O
$molar\_NH_3$	[g mole^{-1}]	molar weights of NH_3

As before, $NH_{3,clean}$=0 and N_2O_{clean}=0 for windrow and home composting. As introduced in chapter 3.3, the emission coefficients are defined as: $NH_{3,clean}$ = 83 %, N_2O_{clean} = 15.

Furthermore, the EASEWASTE model assumes that no degradation of ash and heavy metals occurs during the composting process, while the organic pollutants (DEHP, NPE and PAHs) in the degradable materials fractions are degraded according to the degradation ratios for VS.

The dry matter left after the composting process (ash + non-degraded VS) is distributed between the defined output fractions (i.e. compost, recirculate, wood chips to incineration) using TCs, according to Equation 9 (Boldrin *et al.*, VI):

$$TS_{out,o} = \sum_{1}^{n} TS_{bio,f} \times TC_{o,f} \qquad \text{Equation 9}$$

Where:

$TS_{out,o}$	[kg]	amount of TS in the output -o
$TS_{bio,f}$	[kg]	amount of TS left in waste fraction -f after degradation
$TC_{o,f}$	[%]	TC for fraction -f into output -o

Each of the output flows from biological treatment can be routed to other modules, where further treatment or disposal is modelled and assessed. Compost can be used as a soil amendment on agricultural land or can be used in the preparation of growth media as a substitute for peat materials, while material rejected from the composting process can be sent for thermal treatment.

4.2.2. Use-on-land modelling

The application on land of compost material as soil amendment is modelled in EASEWASTE in a submodel called "Use-on-land". The module, described in details in Hansen *et al*. (2006), was developed to model several key processes taking place in the soil after application of processed organic waste, as shown in Figure 15. In additions, the replacement of mineral fertilizers due to the application of compost is taken into account.

In the land application module, the environmental impacts from compost application are calculated as difference between a reference scenario (i.e. use of mineral fertilizers) and an organic scenario (i.e. application of treated organic waste), using emission coefficients which are specific for the soil type and the crop rotation.

Substitution of mineral fertilizers (usually N, P, and K) is calculated based on the amount of nutrients contained in the compost and using "utilization coefficients" defining the fraction of nutrients utilized by the plants. The substituted process (i.e. production of mineral fertilizer) is selected from an external list of datasets.

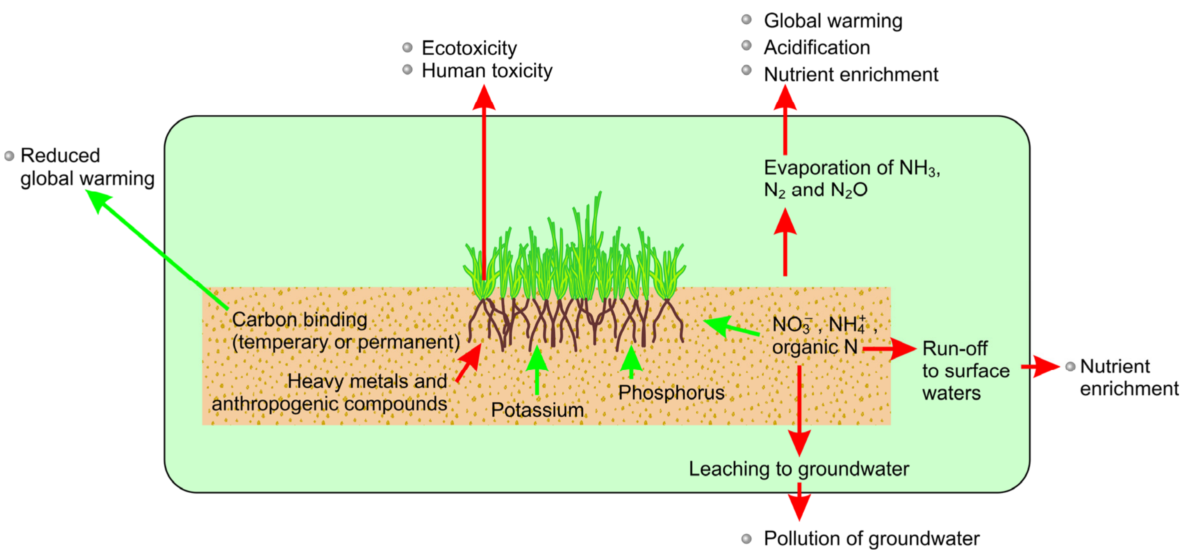

Figure 15 - Environmental impacts from land application of processed organic waste included in the land application submodel in EASEWASTE (from Hansen et al., 2006).

The module is supported with a large inventory of key parameters modelled with the agro-ecosystem model DAISY for the Danish situation, as reported in Bruun *et al*. (2006).

4.2.3. Peat substitution modelling

Compost can be used for diluting peat in growth media preparation. The substitution of peat with compost is normally done on a 1:1 volume basis (Mathur & Voisin, 1996). Avoiding the use of peat means that all environmental burdens occurring during the life cycle of peat can also be prevented. In addition, compost contains some nutrients which can potentially offset the use of mineral fertilizer in growth media. However, the substitution has some drawbacks: in comparison to peat, compost has a higher content of heavy metals and can potentially induce the emission of some gases, such as NH_3 and N_2O.

From a LCA perspective, if system expansion is performed and the two materials provide the same service, the substitution of peat with compost means that the LCI of peat can by "credited" to the environmental profile of compost. The (subtracted) substitution is modelled in EASEWASTE in a specific module called "material utilization".

A typical LCI for peat used in Denmark is presented in Boldrin *et al*. (VII). As shown in Figure 16, such an inventory includes the four main phases of the peat life cycle: 1) preparation and use of peat-land, 2) peat extraction and processing, 3) transportation to the growth media manufacturing plant, and 4) decomposition and environmental effects during and after use on land.

Modelling the use of compost in substitution of peat requires accounting for a number of other aspects. First of all, as established in Boldrin *et al*. (VII), compost and peat have different leaching patterns, mainly because of the higher content of heavy metals in compost. This is taken into account in EASEWASTE by defining different leaching profiles for the two materials.

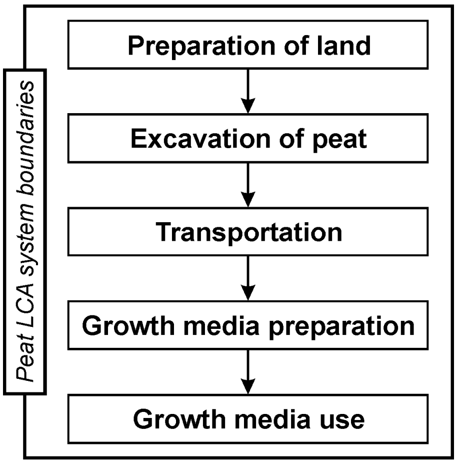

Figure 16 – LCA system boundaries for peat (Boldrin et al., VII).

Secondly, compost contains nutrients, such as N, P, and K, which can potentially offset the use of mineral fertilizers in growth media. This is modelled in the "material utilization" module in EASEWASTE by linking the content of nutrient in compost to external LCI datasets regarding the production of mineral fertilizers.

Thirdly, degradation of compost is slower than peat degradation, meaning that within the 100-year time horizon of the assessment all peat is degraded (Cleary *et al*., 2005), but part of the compost might remain. This is accounted for by defining the fraction of carbon still bound to soil, which in Danish conditions might be of the order of 8-14 % on open land (Bruun *et al*., 2006).

Finally, emissions of gases such as NH_3 and N_2O might occur from the use of compost. Their magnitude has not estimated in the case of compost being mixed in a growth medium. The alternative is to perform a modelling similar to that described for the application of compost on land.

4.2.4. Thermal treatment modelling

Thermal treatment can be applied to garden waste or to the material rejected during the composting process. Thermal treatment can be modelled in EASEWASTE in a dedicated module, described in details in Riber *et al*. (2008). In thic module, each substance is transferred to different outputs by means of TC. In addition, depending on the technology under consideration, process-specific emissions can be defined. If energy recovery is performed, the electricity and/or heat recovered are calculated through the lower heating value of the waste input and the energy recovery factor of the facility, taking into account the fact that part of the energy is used to evaporate the water contained in the waste. Each energy output is linked to a marginal process for energy production, so that the substitution process is modelled.

4.2.5. Life Cycle Impact Assessment (LCIA)

The Life Cycle Impact Assessment (LCIA) phase of an LCA converts physical emissions into potential emissions through characterization factors which are specific for each substance and impact category being considered. In the case study presented later in this thesis, the LCIA was performed based on the EDIP (Environmental Design of Industrial Products) methodology (Wenzel *et al.*, 1997). The potential impact categories relevant for assessment of garden waste and the respective normalization factors are reported in Table 7 (Stranddorf *et al.*, 2005).

Table 7 - Normalisation references for environmental impact categories in EDIP97 (Stranddorf et al., 2005).

Impact category	Geographical scale	Characterisation-unit	Normalization reference [Characterisation-unit $person^{-1}$ $year^{-1}$]
Non-toxic impacts			
Global warming (GW)	Global	kg CO_2-equivalents	$8.7 \cdot 10^3$
Acidification (AC)	Regional	kg SO_2-equivalents	$7.4 \cdot 10^1$
Nutrient enrichment (NE)	Regional	kg NO_3-equivalents	$1.2 \cdot 10^2$
Photochemical ozone formation (POF)	Regional	kg C_2H_4-equivalents	$2.5 \cdot 10^1$
Toxic impacts			
Human toxicity via air (HT)	Local	m^3 air	$6.1 \cdot 10^{10}$
- Human toxicity via water	Regional	m^3 water	$5.2 \cdot 10^4$
- Human toxicity via soil	Regional	m^3 soil	$1.3 \cdot 10^2$
- Ecotoxicity via water	Regional	m^3 water	$3.5 \cdot 10^5$
- Ecotoxicity via soil	Regional	m^3 soil	$9.6 \cdot 10^5$

Considering the high content of biogenic carbon in garden waste, the characterization procedure for CO_2 emissions can be relevant for the results of the LCA study. Within garden waste system boundaries, the important flows of carbon to be accounted for are: CO_2 emissions (both fossil and biogenic) to air, C bound in soil after application of compost on land, and the substitution of energy by heat and power generation at the incinerator. As shown in Table 8, different approaches for assigning GWP to the different biogenic C flows - and converting them to CO_2-equivalents - are used in the literature. As demonstrated in Christensen *et al*. (VIII), the validity of such methods depends on how the waste system boundaries are defined with respect to aspects such as the interaction with the forestry and energy sectors and carbon sequestration. Furthermore, Christensen *et al*. (VIII) conclude that only two characterization methods (reported in Table 9) are consistent in all boundary conditions.

In the EASEWASTE modelling carried out for the present thesis, Criterion 1 is used for characterization of C flows in garden waste management: biogenic CO2 is considered neutral with respect to global warming and carbon binding to soil, after compost application, is credited to the system as saved CO2 emissions. The

Table 8 – Assignment of GWP indices and system boundary specifications in recent LCA-modelling of waste rich in C (from Christensen et al., VIII).

Reference	GWP indices assigning				System boundary specification				
	C to air		C bound		Energy substitution			Wood substitution	
	Biogenic	Fossil	Biogenic	Fossil	Biogenic	Fossil	Other	Degrade	Fossil fuel
Smith *et al.*(2001)[a]	0	1	(−1/0)	0	NU	−1	0	NU	NU
Grant *et al.* (2001)	0	1	(−1/0)	0	NU	−1	NU	NU	NU
Lopes *et al.* (2003)	0	1	NU	NU	0	−1	NU	NU	NU
Dahlbo *et al.* (2005)	0	1	NU	NU	NU	−1	NU	NU	NU
Raymer (2006)	0	1	NU	NU	NU	NU	NU	NU	−1
Schmidt *et al.* (2007)	0	1	−1 (?)	0	0	−1	NU	NU	−1

GWP indices assigning: 1, counted; 0, not counted; −1, subtracted; NU, not used or not relevant in the reference; ?, unclear/not specified.
[a]The study by Smith *et al.* (2001) assumed several scenarios concerning C-biogenic bound or not. Both energy substitution from fossil fuels or from wind have been assessed, but offset against biogenic energy production has not been included.

Table 9 – Consistent assignment of GWP indices and system boundary specifications in LCA-modelling (Christensen et al., VIII).

Criteria	GWP indices assigning				System boundary specification				
	C to air		C bound		Energy substitution			Wood substitution	
	Biogenic	Fossil	Biogenic	Fossil	Biogenic	Fossil	Other	No, it degrades	Fossil fuel
Criteria 1	0	1	-1	0	0	-1	0	0	-1
Criteria 2	1	1	0	0	-1	-1	0	1	0

CO2 emissions from incineration of garden waste are also considered neutral while emissions from the marginal technology for energy production are calculated according to the type of fuel offset (e.g. CO_2 from coal is considered fossil). The reasons for choosing Criterion 1 are two-fold. Firstly, there is no need to expand the boundaries of the assessment into the forestry sector (Christensen *et al*., VIII). Secondly, the criterion is consistent with the IPCC indications for emission inventories in the waste sector (Eggleston *et al*., 2006). However, EASEWASTE makes sure that, even if not counted as a potential contribution to the global warming category, the emissions of biogenic CO_2 are reported in the inventory list.

4.2.6. Material Flow Analysis (MFA) and Substance Flow Analysis (SFA)

The combination of MFA/SFA and LCA is a powerful tool for performing robust environmental assessments of waste management policies, technologies, and implementation mechanisms (Brunner & Ma, 2009). Supporting the EASEWASTE LCA-modelling with MFA and SFA consents to clearly and thoroughly link the inputs and the outputs of the treatment process and provide a transparent representation of the studied system (Brunner & Ma, 2009).

In the present thesis, the mass-balance model STAN (Cencic & Rechberger, 2008) was used for MFA and SFA of the studied scenario. STAN was also used to estimate VS degradation and TS transfer coefficients used in EASEWASTE.

4.3. Perspectives in waste-LCA modelling

The presented EASEWASTE model covers the properties of complexity, user-friendliness, flexibility and transparency expected in a robust waste-LCA model. Comparison is possible at system or technology level, and also on single waste fractions. EASEWASTE constantly tracks the composition of different flows of materials in the system and handles both process- and input-specific emissions from the single processes.

The system boundary definition and the calculation structure permit a proper coverage of temporal and spatial dimensions - adjustable according to the requirements of the study - and distribute the emissions to a large range of potential impact categories. Interpretation is facilitated in the model, because results can be disaggregated at the process level and for each impact category. The transparency of EASEWASTE is further improved by supporting the LCA-modelling with MFA and SFA performed by means of STAN. The holistic approach of the model takes into account downstream benefits ascribable to the waste system, including different substitution processes (e.g. peat, fertilizers, energy, etc…).

4.4. Limitations of waste-LCA modelling

A few aspects limit the comprehensiveness of the modelling and the subsequent results. First of all, as noted in Boldrin *et al*. (VII), the LCA approach does not properly take into account some toxicity aspects of the application of compost on land. In fact, the current LCA methodology calculates the potential toxic effects according to the amounts of specific compounds released to the environment. Since the toxicity of a substance is instead determined by its concentration, such simplification of the LCA-methodology could prompt overestimation of the toxicity impacts. Due to the assumption of linear correlation instead of an S-shaped concentration response curve, this is especially true in those cases where concentrations of pollutants are low.

Secondly, the use of compost on land can potentially improve the quality of the soil where it is applied, but such benefits are not quantifiable in an LCA context. Examples of beneficial effects of compost application are:

- Increased content of organic matter in the soil, which prevents, among other things, desertification and erosion.
- Enhanced hydraulic retention, preventing both floods and droughts.
- Improved workability, reducing the use of heavy machinery in the fields.
- Increased the biological activity, which improves fertility and biodiversity.
- Reduced use of pesticides.

Thirdly, to reflect fully the reality, the assessment should take into consideration (from a consequential-LCA perspective) some specific aspects of the single treatment facilities under consideration, as suggested by Ekvall *et al*. (2007). For instance, the capacity of a treatment plant should be of concern: if waste is diverted from Treatment 1 to Treatment 2, some capacity is made available in Treatment 1. The possibility of using such spare capacity for other purposes should be modelled within the system boundaries. Furthermore, diverting a waste stream from one treatment to another would modify the waste composition and this change might influence the performance of the technology in question. For instance, if the woody fractions of garden waste are diverted to incineration, the feedstock for the composting process might become more compact and gas emission patterns might be different.

Finally, the presented LCA methodology does not properly assess the environmental burdens due to the treatment and/or disposal of wastewater, bottom ash, fly ash, and sludge produced during treatment of garden waste.

5. Environmental assessment of garden waste management

5.1. LCA of garden waste management

An LCA study comprises four interconnected phases: 1) goal and scope definition, 2) inventory, 3) impact assessment, and 4) interpretation (ISO 14040, 2006). At a practical level, the development of a waste-LCA study consists of different activities/phases, methodically carried out with respect to the scope of the study.

The scope definition is the first task encountered when initiating a new LCA study and influences the whole development of the assessment. The way the inventory analysis is performed depends on the scope of the study. The collected data must cover the spatial and temporal boundaries of the assessment: specific studies necessitate specific data collected at a local level, generic assessments require instead generic data covering large geographical areas (Bjarndottir *et al*., 2002). The choice of generic data can easily be a source of great uncertainty. The reason for this is that, even for similar systems, broad ranges of basic data can be found in literature, as shown in Boldrin *et al*. (2009).

Different assumptions and decisions, determining to some extent the reliability of the study, are to be made and taken throughout an LCA study, while collecting data, doing the modelling, and interpreting the results. Since the relationship between the subject of the study and the object of the modelling is often critical (Norris, 2009), an effort is required to make the study as realistic as possible (Christensen *et al*., 2007), including an extensive uncertainty analysis for assessing the reliability of the results.

The interpretation of the results comprises a summary and a discussion of the outcomes of the assessment, based on which conclusions and recommendations are drawn (ISO 14040, 2006). Interpretation is facilitated if the assessment is at the same time complex, specific (i.e. containing few assumptions) and transparent, and the results are presented in a disaggregated form, so that the environmental impacts identified during the assessment can be justified (Ross *et al*., 2002).

Not many LCA studies have assessed the management of garden waste from an environmental perspective, and even fewer include a comparison between composting and alternative options. Inventories of CO_2 emissions arising from composting of garden waste are, for instance, reported in Smith *et al*. (2001) and USEPA (2006). LCIs, also including other types of emissions, can be found in Komilis & Ham (2004), Bjarndottir *et al.* (2002), and Recycled Organics Unit (2006), but none of these studies actually compares composting with alternative treatments. Comparisons between composting and incineration of garden waste are presented in Kranert & Gottschall (2008), and Chapman *et al*. (2009), but solely from a global warming perspective.

5.2. Århus case-study

An environmental assessment of garden waste management alternatives was performed based on data for the city of Århus (Denmark), as presented in the previous sections of the thesis. In 2007, Århus, with its 300,000 inhabitants the second largest city in Denmark, generated approx. 45,000 tonnes of garden waste, 16,220 of which were treated in the local composting plant.

The study assessed six scenarios for handling the 16,220 tonnes of garden waste generated in Århus, evaluating windrow composting, incineration, and home composting as treatment technologies:

- *Scenario 1 - Current management*. Central windrow composting is the treatment for all garden waste collected, producing approx. 10,700 tonnes of compost. Foreign items, large items of wood and screen residues (>25 mm) are incinerated.
- *Scenario 2 - Composting and incineration of rejects*. All screen residues (>8mm) are incinerated.
- *Scenario 3 - Composting and seasonal incineration of waste*. All garden waste received in the winter months (December, January, February) is incinerated. In the remaining nine months of the year garden waste is managed in the central composting plant, according to the usual procedure.
- *Scenario 4 – Maximum incineration of waste*. All garden waste received in winter period, screen residues > 8 mm and large items of wood are incinerated. The remaining waste is composted.

- *Scenario 5 - Home composting*. 25 % of the small fraction is treated with home composting. The remaining waste undergoes central composting.
- *Scenario 6 – Home composting and maximum incineration*. 25 % of the small fraction is treated by home composting. Garden waste received in winter period, screen residues > 8 mm and large items of wood are incinerated, while the remaining waste is composted.

A detailed description of the scenarios can be found in (Boldrin *et al*., IX), while a brief overview of the changes assessed is given in Table 10. Figure 17 presents system boundaries for the current management (Scenario 1) of garden waste in Århus. Similar figures for the remaining scenarios can be found in Boldrin *et al*. (2009).

Table 10 – Overview of amounts diverted to incineration and home composting in analysed scenarios.

Scenario	Treatment	Amount (tonnes)	Fraction diverted
1	WTE (wood) WTE (rejects) Home comp.	501 597 -	
2	WTE (wood) WTE (rejects) Home comp.	501 1,749 -	 Recirculate (>8mm)
3	WTE (waste) WTE (rejects) Home comp.	4,631 440 -	Winter waste
4	WTE (waste) WTE (rejects) Home comp.	4,631 1,276 -	Winter waste Recirculate (>8mm)
5	WTE (wood) WTE (rejects) Home comp.	501 604 3,039	 25% fine fraction
6	WTE (waste) WTE (rejects) Home comp.	4,017 1,035 3,039	Winter waste Recirculate (>8mm) 25% fine fraction

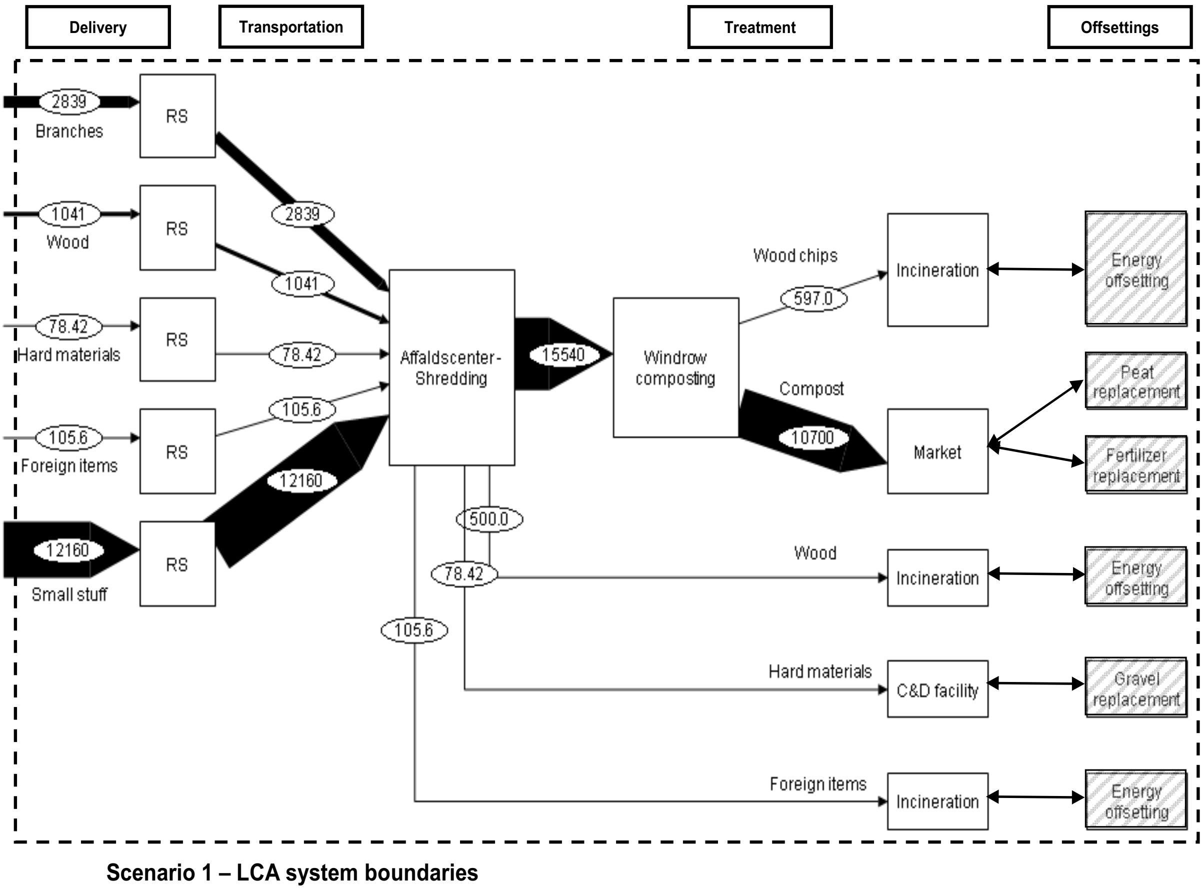

Figure 17 - LCA system boundaries for scenario 1 - Current management of garden waste. Material flows are expressed in tonnes of ww. Taken from (Boldrin et al., IX).

5.2.1. Modelling and assumptions

The time horizon of the assessment was 100 years and system boundaries were expanded to include upstream and downstream processes linked to garden waste management, as shown in Table 11, according to the U-O-D concept.

The assessment was based on the waste composition defined in Section 2. Fuel consumption was estimated to be 8.9 l $tonne^{-1}$ ww of petrol for delivery of waste to the recycling centres (private cars) and 1.52 l $tonne^{-1}$ ww of diesel for its transportation to the composting plant by means of trucks (Boldrin *et al.*, IX). LCIs for the windrow composting facility, incineration, and home composting were based on data presented in Section 3.

Table 11 - Overview of different aspects considered in the assessment (Boldrin et al., IX).

	Indirect: Upstream	Direct: Operation	Indirect: Downstream
Accounted	• Diesel provision. • Electricity provision.	• Combustion of diesel for collection and transportation of garden waste. • Composting plant: - Gas emissions (CO_2-biogenic; CH_4; N_2O, CO, NH_3); - Combustion of diesel. • WTE plant: - Use of materials and energy needed for the combustion process; - Gas emissions from the stack. • C&D facility: - Combustion of diesel. • Home composting: - Gas emissions (CO_2-biogenic; CH_4; N_2O, NH_3).	• Peat substitution: - Substitution of peat; - CO_2-biogenic from compost degradation; - C binding in soil; - N_2O from use-on-land; - Substitution of inorganic fertilizers. • Energy recovery in WTE plant: - Substitution of electricity; - Substitution of heat. • Material recovery in C&D facility: - Substitution of gravel and crushed rock extraction.
Non-accounted	• Construction of treatment facilities and/or machineries. • Provision of other materials. • Construction of plastic composters.	• Composting plant: - Any trace gas release; - Treatment of collected leachate. • Treatment of residues from WTE plant.	• Improved soil quality from use-on-land of compost.

The LCA modelling was performed by means of EASEWASTE and according to the principles presented in Section 4: peat substitution was calculated on a 1:1 volume basis, while mineral replacement was modelled according to the nutrient content of the compost, reported in Andersen *et al*. (III) (see chapter 3.2.2). Combining these values with the utilization rates for the nutrients (assumed to be 20% for N and 100 % for P and K), the utilization of one tonne replaces 1.64 kg N, 1.08 kg P, and 10.8 kg K. A user survey[4] was carried out to estimate the actual substitution rate of peat with compost in gardens: preliminary results indicate that

[4] The user survey was carried out at the recycling stations in Århus in August 2008 and June 2009. The focus of the questionnaire was the substitution ratios of compost and peat. It included questions about alternative soil improving products (peat, sphagnum, fertilizers, manure, etc.) used in the garden, and an estimation of the amount saved when compost is used instead. About 80 interviews were completed during the two events. The survey is described in detail in Boldrin *et al*. (2009).

less than 50 % of the compost used in gardens actually replaces peat. This aspect was taken into account in EASEWASTE by halving the amount of peat replaced (131.5 kg peat instead of 263 kg per tonne of compost). Carbon binding to soil within the 100-year horizon of the assessment was also included. The value assumed in the modelling was 14 % of the carbon contained in compost, taken from the modelling of Bruun *et al*. (2006) for the application of compost on loamy soil in Danish conditions.

With regard to thermal treatment, coal-based electricity and coal-based heat are the marginal technologies for the energy production substituted by the incinerator.

5.2.2. Results

The normalized results presented in Figure 18 show that, in a garden waste system where composting is the main treatment, the composting facility is the major contributor to non-toxic potential impacts (Boldrin *et al*., IX). Emissions of Greenhouse Gases (GHGs) contributing to global warming category occur in different processes: fossil CO_2 is produced from fuel combustion in heavy machinery (e.g. front loaders, excavators, shredders, etc.) and cars (used for waste delivery), while CH_4 and N_2O develop from windrows during composting. Potential impacts on photochemical ozone formation originate because of Volatile Organic Compounds (VOCs), NO_x and CO emissions during fuel combustion in engines. The composting process is also the main contributor to nutrient enrichment and acidification categories. The first is mainly ascribable to NO_x emissions from fuel combustion in heavy machinery and ammonia (NH_3) evaporating from composting windrows. The latter is due to NO_x, NH_3, and SO_2 (from engines).

The results also show that use of compost in replacement of peat ("Use of compost in gardens" in Figure 18) can potentially save emissions of fossil CO_2 from peat degradation and large savings of global warming impacts can be accounted to the system.

Despite the relatively small amount being incinerated, thermal treatment gives credits to the system through electricity and heat produced.

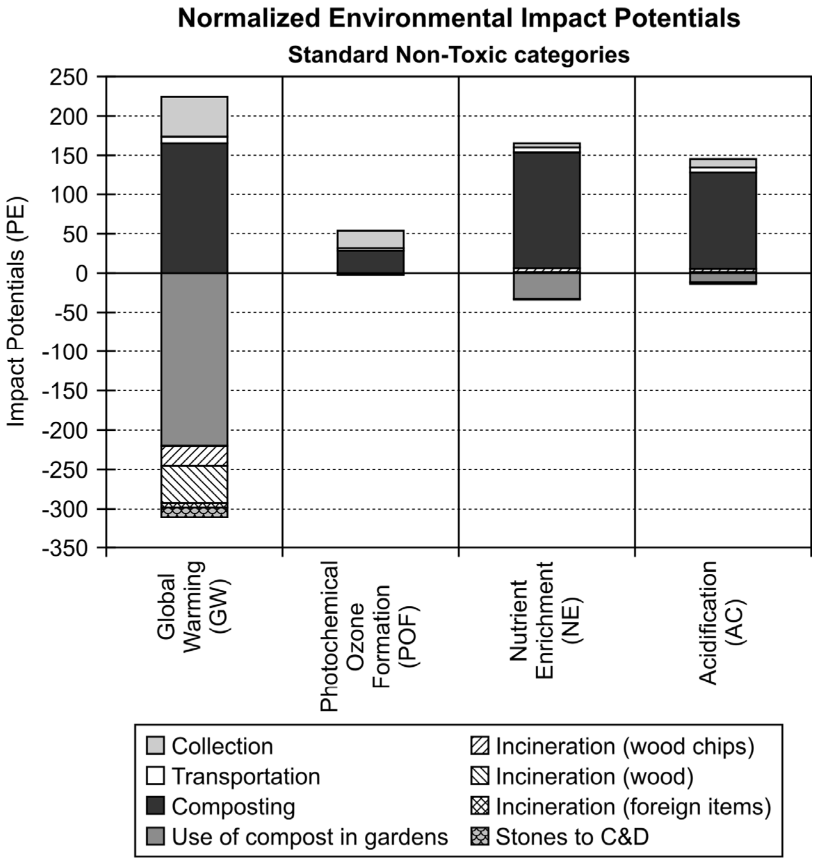

Figure 18 - Potential non-toxic environmental impacts from the current management of garden waste (16,220 tonnes). Taken from Boldrin et al. (IX).

The results for the toxic impact categories (Figure 19) are mainly determined by the metals content – mainly chromium and arsenic - in compost being used in gardens. Such metals are potentially leaching and contribute to both human toxicity categories.

Comparative results for the six analysed scenarios are presented in Figure 20 and Figure 21 for non-toxic and toxic impact categories respectively. It can be seen that both the analysed treatment alternatives – incineration and home composting – for waste diversion result in improvements to the current management (Scenario 1).

Additional incineration of garden waste results in potential extra savings in the global warming category from avoided production of electricity and heat from fossil fuels (coal). Such improvements, in the order of 26 to 208 PE (229 to 1814 tonne CO2-eq.) depending on the amount of waste, add to the savings already

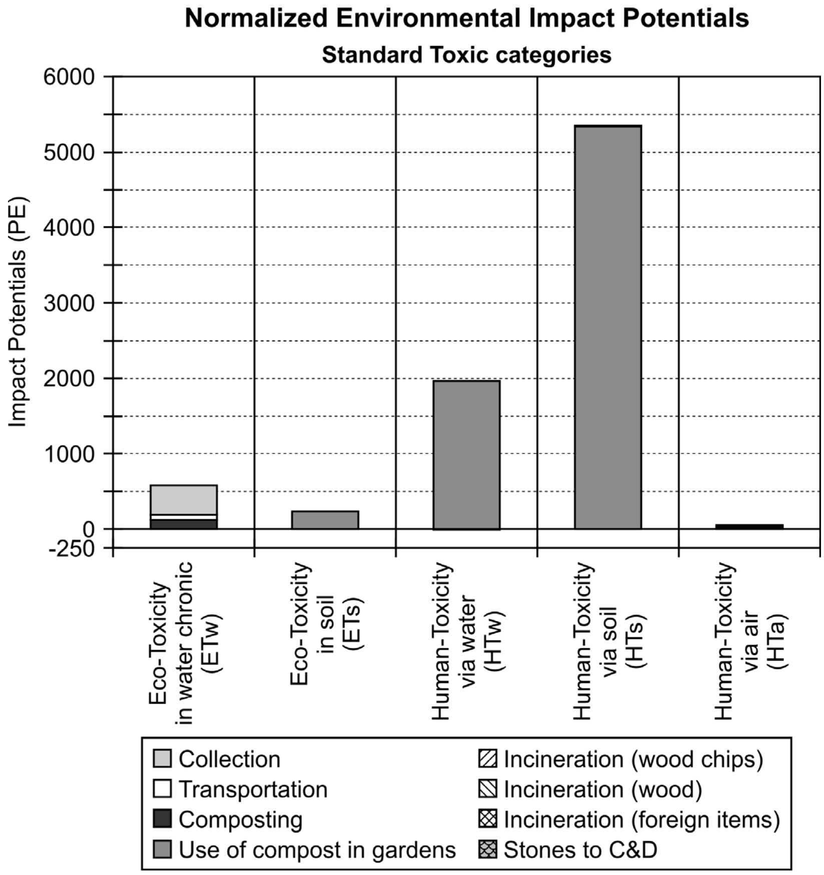

Figure 19 - Potential toxic environmental impact from the current management of garden waste (16,220 tonnes). Taken from Boldrin et al. (IX).

accounted for in the current system (87 PE or 757 tonne CO_2-eq.). On the other hand, greater emissions of NO_x are produced with increased incineration, worsening the environmental profiles of acidification and nutrient enrichment categories.

It is worth remembering that, as mentioned in Section 2, the amount of garden waste that could be optimally diverted to incineration is limited. For technical reasons, the ash content and the LHV impose some restrictions relative to an optimal incineration process. Material with suitable properties includes wood and large branches separated from the mixed garden waste or the unsorted garden waste collected in the winter months, when woody materials are dominant (Boldrin *at al.*, 2009). It should be noted that, compared to the analysed scenarios, additional diversion of high energy fractions of garden waste is potentially possible. However, this would require the introduction of schemes capable of sorting out woody materials which could be sent to incineration, while

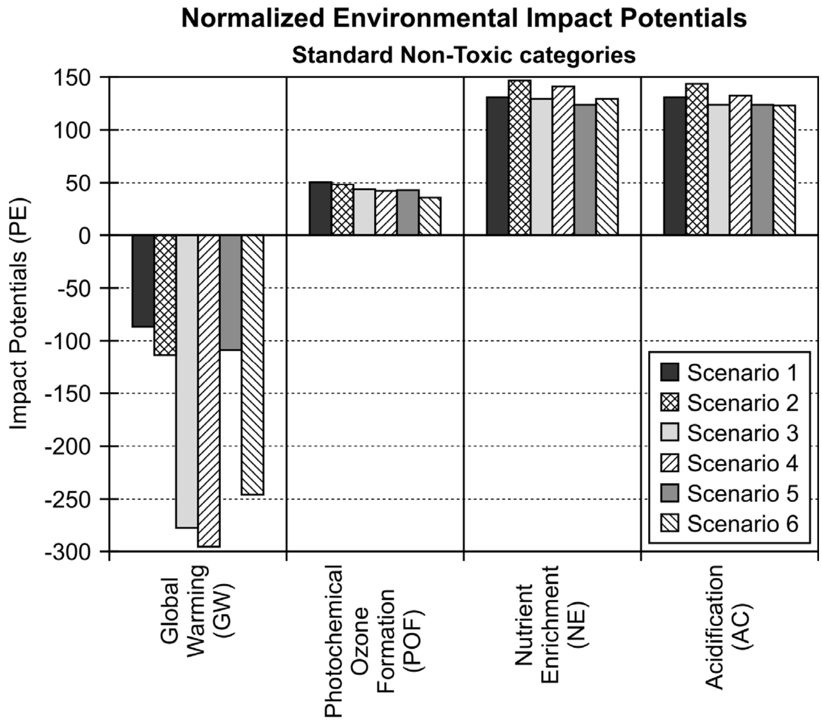

Figure 20 – Comparison of potential non-toxic environmental impacts for analyzed scenarios (16,220 tonnes of garden wastes). Taken from Boldrin et al. (IX).

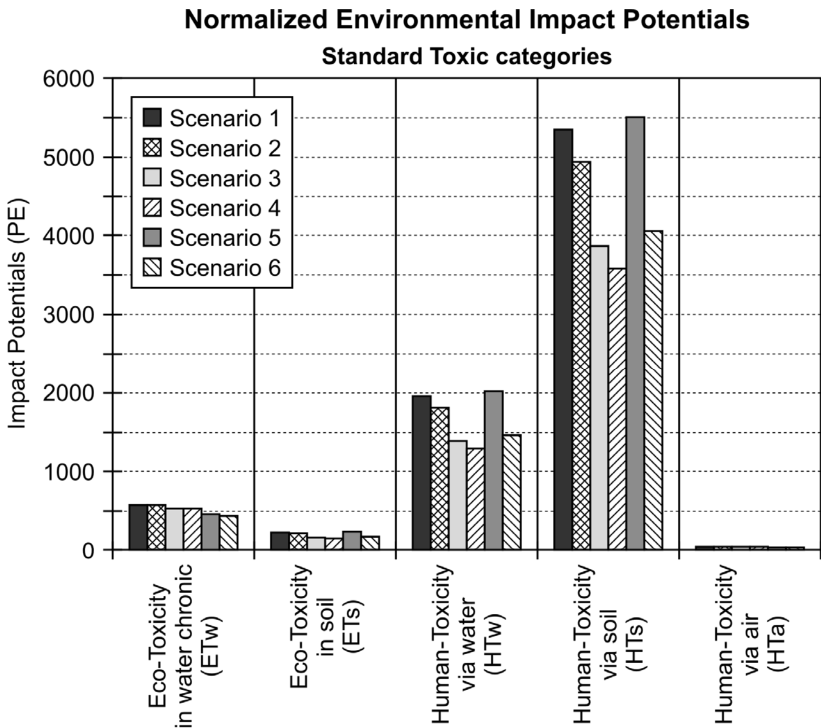

Figure 21 – Comparison of potential toxic environmental impacts for analyzed scenarios (16,220 tonnes of garden waste). Taken from Boldrin et al. (IX).

the remaining fine fraction could be sent to composting. For example, citizens could be asked to sort garden waste in small and large items when delivering garden waste at the recycling centre. Another alternative could be the use of a trommel screen with large mesh-size (e.g. 500 mm) prior to the shredding operations performed at the composting plant.

Introduction of home composting (Scenario 5 and 6) has some benefits in all non-toxic categories, mainly because of the avoidance of waste collection by means of private cars. Similar to incineration, the amount of garden waste potentially treatable by home composting is relatively limited, because of space availability in backyards and the size of the materials (large wooded items are too big for small plastic compost bins).

5.3. Uncertainty analysis

As suggested in ISO-standard 14040 (2006), an analysis of uncertainty should be performed on the LCA study. Uncertainty analysis is a systematic characterization and quantification of the uncertainty introduced in the results of the LCA study as a consequence of assumptions, model imprecision, input uncertainty and data inconsistency. The purpose of the uncertainty analysis is to provide information useful to define the correctness of the conclusions of the study (Norris, 2009). Uncertainty analysis can be performed in a quantitative or in a qualitative way. A statistical quantification should be performed whenever sufficient data are available. If quantification is not possible, a qualitative description is still useful for explaining the relevance of indicator results (Ross *et al.*, 2002), so that the most influential uncertainties are recognized and screening solutions for improving data reliability can be identified and implemented back into the study (Bjarnadottir *et al.*, 2002; Norris, 2009). However, despite the fact that it defines the reliability of the results and improves the credibility of the study, the uncertainty analysis does not reduce the uncertainty in itself.

In the Århus case study, the accuracy of the results was evaluated by means of “uncertainty importance” analysis (Björklund, 2002), for determining how the total uncertainty of the results depends on the uncertainty of different factors. Key parameters were selected and a qualitative description of their accuracy determined. This information was then combined with the results of a sensitivity analysis, so that the uncertainty importance for the specific parameters could be determined.

The selected parameters and a qualitative evaluation of their relevance on the results of the assessment – according to section 5.2 - are reported in Table 12.

Table 12 – Relevance of selected parameters on the results of the assessment.

Parameter	Relevance on the results
Peat substitution ratio	Large
CH_4 emissions during composting	Medium
N losses during composting	Medium
Collection distance	Small
Marginal electricity mix	Large

A qualitative evaluation of the uncertainty related to each parameter was described.

- Peat substitution ratio. The utilization correction factor (50%) used for peat replacement is considered highly uncertain, because it is based on a preliminary user survey with a limited number of responses.
- CH_4 emissions during composting. Measurements were precise and repeated, estimates are in accordance with the carbon balance. The uncertainty of this parameter is considered to be low.
- N losses during composting. Both ammonia measurements and the N balance were inaccurate, so the N degradation (8% of the input N) value is considered highly uncertain.
- The driven distance for garden waste delivery was estimated through the user survey and results were in accordance with previous studies. Medium uncertainty is associated with this parameter.
- Marginal energy mix was assumed according to previous studies on the specific Århus incinerator (Riber *et al.*, 2008). This assumption is considered to have low uncertainty.

A sensitivity test, aiming to determine how a result is influenced by a parameter, was performed by varying the different parameters in selected scenarios (1 & 4), as specified in Table 13. The quantitative results of the test are shown graphically in Figure 22 and Figure 23 by means of variation intervals, showing the consequences of the changes mentioned. A qualitative indicator describing the sensitivity of each parameter relative to the different impact categories was then defined according to the following considerations. For a specific impact category, high sensitivity was assigned if the variation interval was large relatively to the impact in itself or if the variation interval was larger than the absolute (numerical) difference found between the analysed scenarios. The results are reported in Table 14.

Table 13 – Sensitivity test for different parameters and scenarios (Boldrin et al., IX).

Test name	Tested scenario	Parameter changed	Change	From	To	
Scenario 1 – peat	Scenario 1	Peat substitution	± 40 % (± 20 %)	131.5 kg (50%)	79 kg (30 %)	184 kg (70 %)
Scenario 1 – methane	Scenario 1	CH_4-C emissions	± 50 %	2.24 %	1.12 %	3.36 %
Scenario 1 – N balance	Scenario 1	N degradation	± 50 %	8 %	4 %	12 %
Scenario 1 – cars	Scenario 1	Gas. consumption	± 50 %	8.9 l/km	13.4 l/km	4.4 l/km
Scenario 1 – energy	Scenario 1	Marginal electricity mix		Coal	Av. Danish mix	
Scenario 4 – energy	Scenario 4					

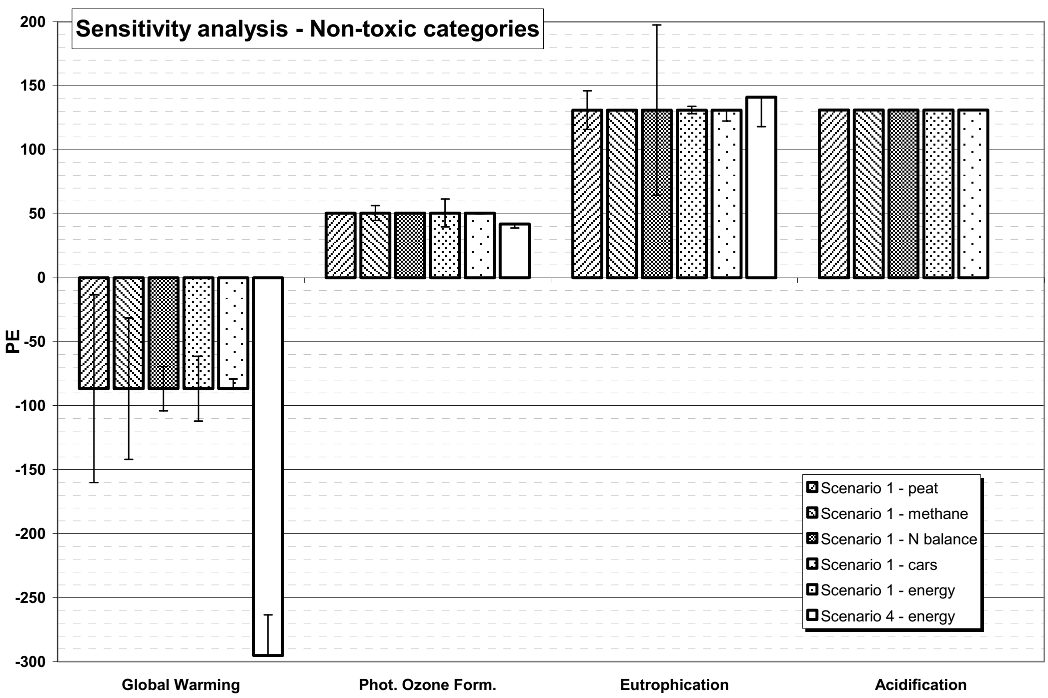

Figure 22 – Results of the sensitivity test for non-toxic impact categories (Boldrin et al., IX).

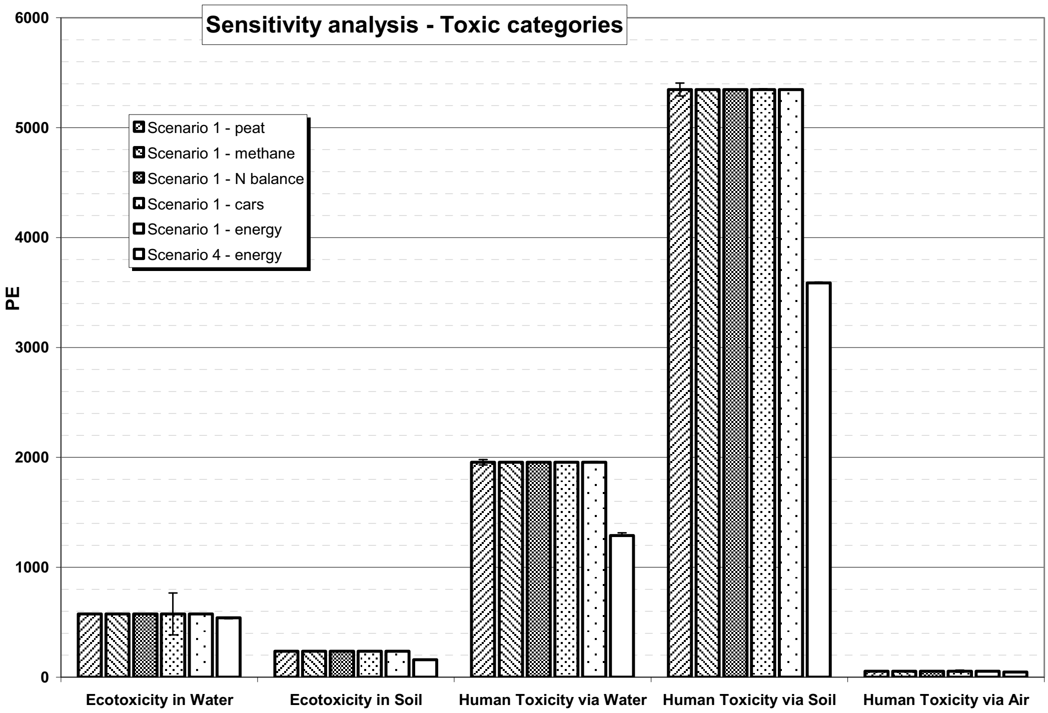

Figure 23 – Results of the sensitivity test for toxic impact categories (Boldrin et al., IX).

The qualitative results of the uncertainty importance analysis are shown in Table 14, defined combining the results of the relevance analysis, the uncertainty analysis and the sensitivity analysis. These results indicate that the most problematic parameters are peat substitution and nitrogen degradation rate, but they also reveal that, even in the worst-case scenario, the conclusions of the assessment might not be altered (Boldrin *et al*., IX; Boldrin *et al*., 2009).

Table 14 - Results of the qualitative uncertainty importance analysis (Boldrin et al., IX).

Parameter	Relevance on the results	Uncertainty	Sensitivity	Uncertainty importance
Peat substitution ratio	Large	High	GW: medium NE: high	High
CH_4 emissions during composting	Medium	Low	GW: medium	Low
N losses during composting	Medium	High	AC, NE: High	High
Collection distance	Low	Medium	GW,AC,HT: medium POF,ET: high	Small
Marginal electricity mix	High	Low	AC,NE: medium HT: high	Medium

5.3.1. Missing aspects

Modelling the application on land of compost in EASEWASTE is based on a rather complex agro-ecological modelling of biological and physicochemical processes occurring in the soil, which takes into account several factors, such as hydrogeological parameters (i.e. soil temperature, evapotranspiration, and soil water transport), the nitrogen cycle, and soil organic matter mineralization, including C and N dynamics (Hansen *et al*., 1991). In addition to this, aspects of compost management - application (i.e., silviculture, horticulture, agriculture, and landscaping), application rate, regional and local climatic factors, and soil type (USEPA, 2006) - contribute to the complexity of the modelling. Such factors can actually be included in the modelling, but whether the chosen parameters are representative enough for describing very local conditions could be open to discussion when validating the study.

Another potentially occurring aspect, not included in the assessment, is the excessive application of compost in gardens, with consequent detrimental effects. Possible excessive application could be noted in the preliminary survey, where it was seen that fewer than 50 % of the people used less growth medium when also using compost. This indicates a redundant use of compost and growth medium at the same time. The excessive application was not included in the assessment because it was not possible to define its extent, which depends, among other things, on the amount of compost purchased, the dimension of the gardens where the compost is applied and the type of vegetation grown.

The treatment of wastewater, generated in the different processes, was not included in the assessment because of lack of data. If, on one hand, the wastewater collected in the composting facility could represent a marginal burden, on the other hand the wastewater generated in the WTE plant might contain rather large loads of pollutants requiring removal.

6. Discussion and conclusions

This thesis and the activities included in it have demonstrated an operative approach for the execution of robust environmental assessments of garden waste management based on LCA methodology. Different activities were carried out in order to reduce methodically the uncertainty emerging in different phases of a waste-LCA study. The approach adopted ensured that the desirable features of an LCA were achieved.

The Life Cycle Inventory phase must provide data which are specific and precise enough to fulfil the scope of the study and capable of being modelled without major assumptions having to be made. In the present thesis, data collection covered all aspects included in the system boundary of the assessment and accurate inventories were established. Furthermore, specific sampling methods for waste composition and gas emissions were developed and validated. These methods are low-cost and can be performed on site, meaning that they are applicable and repeatable at other composting sites of interest.

The waste-LCA model EASEWASTE proved to be a proper platform for completing LCA studies of the necessary complexity and transparency, so that the results are both credible and comprehensible. Specific modelling for the replacement of peat with compost was developed and the resulting benefits – in particular in terms of global warming - could be credited to the system.

The case-study showed that, if the necessary complexity and transparency are provided, LCA is a useful tool not only for comparing different scenarios but also for providing useful insights into the analysed system, enabling it to be improved. In this context, the use of MFA in support of LCA proved to be a powerful tool for an understanding of the system studied and for performing an exhaustive uncertainty analysis. It also proved to be helpful when drawing conclusions. Furthermore, the case study showed that, when performing an LCA, the modelling approach should be both consistent with respect to the subject of the modelling and flexible so that different aspects can be taken into consideration. This proved to be particularly true with regard to peat replacement: a “simple” substitution modelling was found to be insufficient and aspects of people’s behaviour had to be taken into account.

6.1. Environmental aspects of garden waste management

The Århus case study showed that a garden waste management system based on windrow composting generates rather small potential impacts on the environment. These impacts are in the order of a few mPE per tonne of waste treated: -1 to 0.5 mPE tonne^{-1} of ww for the non-toxic categories and up to 18 mPE tonne^{-1} of ww for the toxic categories. This is several orders of magnitude smaller than that found for other types of municipal solid waste (Kirkeby *et al.*, 2006b). Furthermore, the chemical analyses showed that the compost produced from garden waste contains low amounts of contaminants and it is suitable for organic farming (Andersen *et al.*, III).

Clear differences between the analysed scenarios were found for some impact categories, indicating that the diversion of waste to alternative options should be taken into consideration. Among the non-toxic categories, results for the potential impacts on global warming revealed that both incineration and home-composting might improve the performance of the current practice. In the first case, the main reason is the energy recovery from the waste, offsetting energy produced from a fossil source such as coal. The marginal technology employed is, in this case, a crucial assumption for the outcome of the assessment. Regarding home composting, the gained benefits are due to the avoidance of delivery of waste by private cars, but no major improvements were found.

The results for the toxic categories show relatively high potential impacts on human toxicity, the key factor being the content of heavy metals in the compost spread on land. However, rather than raising a major environmental concern, such results reveal that the LCA methodology is probably overestimating the impacts from spreading of compost on soil. The reason for this is that the LCA methodology estimates the potential toxic effects based on the amount of heavy metals, without taking into account effective concentrations. Seen from a different perspective, most of the heavy metals contained in compost originate from the soil fraction contained in garden waste (Boldrin & Christensen, I) and therefore do not contribute to an increase in the background concentration of heavy metals in the soil once compost is used in gardens.

Utilization of compost in gardens in substitution of commercial growth media has potential benefits for the environment, representing a major credit to the composting system. However, the user survey indicates that in many cases such benefits are not fully achieved due to the user behaviour.

6.2. Recommendations

The environmental assessment indicates that current practices for garden waste management could be made more environmentally sound if composting operations were optimized. For instance, a relevant environmental load is represented by GHG emissions (CH_4 and N_2O) occurring during degradation of organic materials in windrow composting. Possible adjustments, regarding for instance the turning frequency of the heaps or the size of the windrows, should be investigated to determine whether such gas emissions can be prevented.

Given that high energy recovery is performed, thermal treatment of some garden waste showed potential environmental benefits. However, various technical aspects determine that only garden waste with specific characteristics (e.g. high LHV and low ash content) can be optimally incinerated. The Århus case study showed that wherever sorting of waste is feasible, incineration of the woody fractions will result in large benefits. If, instead, mixed garden waste is to be incinerated, diversion should be considered on a seasonal basis and waste collected during winter months utilized.

Use of compost in gardens was found to be very beneficial for the system, but the preliminary user survey revealed that probably less than 50% of the potential benefits are actually achieved. An effort should be made to increase and/or optimize the use of compost in substitution of commercial growth media, possibly by means of education campaigns aimed to explaining to people the benefits and correct utilization of compost.

The case-study regarding Århus Municipality covered a broad range of relevant aspects for the management of garden waste. The modelled aspects – waste composition, windrow composting, incineration, use of compost in gardens – might reflect "typical" conditions for the Danish situation. If this case is verified, the results of the study indicate that a strategy based on diversion of a part of garden waste to energy production is favourable. In such case, schemes for sorting woody materials from garden waste need to be implemented. Different solutions might be possible, including sorting schemes at recycling centres or screening prior to shredding at the composting plant.

6.3. Further research

A few issues raised in this thesis necessitate further investigation.

First of all, the characterization of garden waste should be repeated, eventually at different locations, so that a comparison with the results presented here is possible.

From an inventory perspective, a more accurate method for NH_3 measurement should be developed. A precise quantification of NH_3 emissions will improve the accuracy of the N balance and consequently a validation of N_2O measurement will be possible. Furthermore, different aspects of home composting, including gas emissions and leaching, should be investigated so that a complete evaluation can be performed.

Some flows of material were not covered in the assessment. An attempt should be made to resolve these gaps in the data, which include treatments of wastewater generated in the composting facility and in the incinerator, and the solid residues – bottom ash, fly ash, APC residues, and sludge - produced during thermal treatment.

It is necessary, moreover, to develop further the LCA methodology in order to include correctly environmental aspects of utilization of compost on land. Firstly, balanced characterization factors should be defined regarding the toxicity of heavy metals in soil, taking into account their concentrations and the thresholds of specific compounds rather than only their amount. Secondly, a methodological framework for including the potential benefits on the soil quality of compost application should be established. Accounting aspects such as the increased content of organic matter, reduced need for pesticides, enhanced hydraulic retention, and improved workability, could improve the environmental performance of systems based on composting. This is especially true if the assessment relates to regions of poor soil quality.

Further investigations should also be carried out in order to understand fully how compost is used in gardens. In particular, the behaviour of people when using compost needs to be defined, so that realistic modelling of replacement patterns or inefficient usage can be performed.

An additional scenario to be investigated is the use of garden waste – or part of it – for production of Residual Derived Fuel (RDF) and subsequent co-combustion in coal power plants or combustion in dedicated RDF incinerators. The advantage of this option is that RDF has a higher energy content than wet waste, making it suitable for incineration technologies with higher efficiency for electricity production. A drawback of this solution is the energy requirement for RDF production. The assessment of an RDF scenario would require identifying proper technologies for RDF production and combustion and defining the environmental aspects related to their use.

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8. Appendices

I. Boldrin, A. & Christensen, T.H. 2009. Seasonal generation and composition of Danish garden waste. Submitted to *Waste Management*.

II. Boldrin, A., Spliid, H. & Christensen, T.H. 2009. A novel approach for representative sampling of garden waste. Submitted to *Science of the Total Environment*.

III. Andersen, J.K, Boldrin, A., Christensen, T.H. & Scheutz, C. 2009. Mass balances and life cycle inventory for a garden waste windrow composting plant (Aarhus, Denmark). Submitted to *Waste Management & Research*.

IV. Boldrin, A., Andersen, J.K., Møller, J., Favoino, E. & Christensen, T.H. 2009. Composting and compost utilization: Accounting of greenhouse gases and global warming contributions. *Waste Management & Research*, 27, DOI: 0734242X09.

V. Andersen, J.K, Boldrin, A., Samuelsson, J., Christensen, T.H. & Scheutz, C. 2009. Quantification of GHG emissions from windrow composting of garden waste. Submitted to *Journal of Environmental Quality*.

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VII. Boldrin, A., Hartling, K.R., Laugen, M.M. & Christensen, T.H. 2009. Use of compost and peat in growth media preparation: an environmental comparison using LCA-modelling (EASEWASTE). Submitted to *Resource, Conservation and Recycling*.

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IX. Boldrin, A., Andersen, J.K. & Christensen, T.H. 2009. Environmental assessment of garden waste management in the Municipality of Århus (EASEWASTE). Submitted to *Environmental Science & Technology*.

These papers are included in the printed version of the thesis but not in the www-version. Copies of the papers can be obtained from the Library at the Department of Environmental Engineering, DTU (library@env.dtu.dk).

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Water Resource Engineering, Urban Water Engineering,
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The department dates back to 1865, when Ludvig August Colding, the founder of the department, gave the first lecture on sanitary engineering as response to the cholera epidemics in Copenhagen in the late 1800s.

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